INTERTEMPORAL INCENTIVES AND MORAL HAZARD IN NONPOINT-SOURCE POLLUTION

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1. Introduction

The inability to measure with sufficient accuracy the emissions of individual dischargers in nonpoint-source pollution renders inadequate the standard instruments of environmental policy (Pigouvian taxes, controls, etc.) as a means of inducing dischargers to follow socially desirable policies. Measurement of ambient pollution concentration at some “receptor point” without any possibility of inferring individual emissions is a source of moral hazard. Dischargers will choose higher than socially desirable emission levels if by doing so they can increase their profits.

This type of moral hazard problem can be prevented by introducing incentive schemes that include penalties, rewards, or some combination of both that depend on deviations between desired and measured ambient concentration levels (Meran and Schwalbe 1987, Segerson 1988, Xepapadeas 1991). These schemes, however, are essentially static since they ignore the dynamic process of the pollutant accumulation and the effects it might have on individual behavior if it is imposed, through some incentive scheme, as a restriction on the dischargers’ intertemporal profit maximization problem.

The purpose of this paper is to explore the possibilities of designing intertemporal incentive schemes that would induce dischargers to follow a policy resulting in a socially-desirable long run equilibrium concentration level. In this pollutant context, differentiable incentive schemes depending on deviations between desired and observed ambient levels and the social dynamic shadow cost of pollutant concentration are examined under conditions of certainty and uncertainty. In the latter case, uncertainty is associated with random natural pollution decay rate and results in incentive dependence on the mean and variance of the pollutant concentration along the socially optimal path. The problem is analyzed in a differential game framework. Incentives corresponding to open-loop, feedback and perfect conjectural Nash equilibrium (Basar and Olsder 1982, Fershtman 1987, Fershtman and Kamien 1985) are formulated. The resulting schemes are not in general similar to the static ones, by which optimal behavior can be secured by penalizing each discharger with the full social cost of deviations from desired ambient levels (Segerson 1988, Xepapadeas 1991). As a result, adoption of static rules might lead to socially inefficient outcomes in the long run.

The paper is organized as follows. The second section analyzes the path of pollutant accumulation resulting from the objectives of maximizing either profit or some welfare indicator that accounts for damages due to environmental pollution, using deterministic and stochastic specifications. The long run equilibrium accumulation levels of the pollutant under the different objectives are determined and compared. In the third section, the efficient incentive schemes are constructed under different assumptions about the information structure that the discharging firms use to determine their strategies with respect to their emissions, and the stochastic properties of the model. Efficient schemes are determined as unit charges on deviations between observed and desired pollutant accumulation levels. Comparisons of the equilibrium pollutant accumulation levels under static and dynamic incentives are also performed. The last section provides some concluding remarks.

2. Long run pollutant accumulation

2.1 Model description

We consider a market consisting of i=1,...,n firms that produce a homogeneous product. output production generates pollution, that can be abated by using additional resources. The benefit of the ith firm at each instant of time can be written as a function of its discharges (Malik 1990)

$$B_i = B_i(e_i(t)),\ B_i' > 0,\ B_i'' < 0 \quad \text{for all } t \in [0,\infty) \tag{1}$$

where $e_i \in E_i \subset R_+$ denotes discharges of the ith firm and E_i is. assumed to be compact and convex. [1]

Let $x \in X \subset R_+$ with X compact and convex, denote the ambient concentration of the pollutant generated by the firms' productive activities. It is assumed that environmental uncertainty associated with factors like weather or topographical conditions result in random natural pollution decay rate (Plourde and Yeung 1989, Xepapadeas 1990). The evolution of the pollutant concentration can be described by the stochastic differential equation

$$dx(t)=[\Sigma_i e_i(t)-bx(t)]dt+\sigma x(t)dz(t),x(0)=x_0 \text{ non-random} \qquad (2)$$

where {z(t)} is a Wiener process. [2]

In (1), $\sum e_i-bx$ is the instantaneous expected change of x per unit of time, with (-bx) being the mean of the natural decay process, assuming exponential natural pollution decay rate, and $(\sigma x)^2$ is its instantaneous variance. Hence the accumulation of the pollutant follows a diffusion process. The effects of the pollutant accumulation are described by a damage function

$$D=D(x(t)),\ D'>0,\ D''>0,\ \text{for all } t\in[0,\infty) \qquad (3)$$

The upper bound assumption on the pollutant accumulation means that $D(x)\to\infty$ as $x\to x_{max}$ (Kamien and Schwartz 1982). Relations (1) - (3) can be used to analyze the long run pollutant accumulation resulting from profit maximization or from the maximization of some welfare indicator.

2.2 Profit maximization

The ith firm will choose the discharge level that maximizes (1), resulting in first order conditions

$$B_i'(\hat{e}_i)\le 0,\hat{e}_i\ge 0 \quad i=1,\ldots,n \quad all\ t\in[0,\infty) \qquad (4)$$

The dynamics of environmental change, due to profit maximizing behavior, are obtained by substituting $\hat{e}_i$ into (2). To simplify things at a first stage, consider the certainty case $\sigma=0$. From (2) we obtain

$$\dot{x}=\Sigma_i \hat{e}_i - bx \qquad (5)$$

The long run pollutant accumulation corresponding to profit maximizing behavior is the equilibrium point $(x_\infty) \in X$ such that $\dot{x}=0$. From (5) we obtain

$$x_\infty = (\textstyle\sum_i \hat{e}_i / b)$$

Since b>O, the equilibrium point is globally asymptotically stable.

When the uncertainty case is examined, the evolution of the pollutant concentration is determined by the solution of (2) with $e_i=\hat{e}_i$ all i and t. This stochastic differential equation has a unique solution as a diffusion process with drift coefficient $(\sum_i \hat{e}_i - bx)$ and diffusion coefficient $(\sigma x)^2$[3]. The diffusion property implies that given some pollutant accumulation x at time t, the probability that in some future time the pollutant accumulation will fall within the interval (a_1, a_2), $0<a_1<a_2<\infty$ can be determined.

<u>Proposition 2.1 Let</u> $(x_m)_\infty$ denote the expected long run pollutant concentration , then $(x_m)_\infty = x_\infty$.

<u>Proof</u> By taking expected values in the integral form of equation (2), using the properties of stochastic integrals (Karlin and Taylor 1981), and denoting $E(x)=x_m$ we obtain the following differential equation for x_m (Gard 1988, theorem 4.5)

$$\dot{x}_m = \Sigma \hat{e}_i - bx_m \ \text{with}\ x_m(0)=x_o$$

which has equilibrium solution

$$(x_m)_\infty=\Sigma\hat{e}_i/b=x_\infty \qquad \blacksquare$$

Thus, in the long run the expected pollutant concentration is the same as in the deterministic case. [4]

2.3 Social optimum

A social planner will seek the discharge levels that maximize expected net benefits, that is

$$\max_{e_i} E_0\int_0^\infty \exp-\rho t\,[\Sigma_i B_i(e_i)-D(x)]\,dt$$

subject to (2) and $e_i \in E_i$ all $t\in[0,\infty)$ and i=1,...,n (P. 1)

where p>0 denotes the discount rate.

The cases of certainty and uncertainty are examined in the following.

(i) Certainty, $\sigma=0$

The current value Hamiltonian for problem (P.1) can be written as

$$H(x,e_1,\ldots,e_n,\lambda)=\Sigma_i B_i(e_i)-D(x)+\lambda(\Sigma_i e_i-bx) \qquad (6)$$

where $\lambda(t)$ can be interpreted as the dynamic social shadow cost of pollutant concentration.

The necessary and sufficient conditions for optimality, since H is concave in x, e_i (Seierstad and Sydsaeter 1987), are

$$\frac{\partial H}{\partial e_i}=B_i'(e_i^*)\leq-\lambda\ ,\ e_i^*\geq0\quad i=1,\ldots,n \qquad (7.1)$$

$$\dot{\lambda}=(\rho+b)\lambda+D'(x) \qquad (7.2)$$

equation (2) with $\sigma=0$, and the Arrow type transversality conditions.

From (7.1) we obtain for interior solutions

$$e_i^* = e_i(\lambda) \qquad (8)$$

with $e_i' = -1/B_i'' > 0$ by the implicit function theorem. Thus for $\lambda < 0$, a reduction in absolute value of the social shadow cost of the pollutant will increase discharges.

Substituting (8) into (2) and using (7,2), we obtain the Modified Hamiltonian Dynamic System (MHDS)

$$\dot{x} = \Sigma_i e_i(\lambda) - bx \qquad (9.1)$$

$$\dot{\lambda} = (\rho + b)\lambda + D'(x) \qquad (9.2)$$

The properties of the long run equilibrium for the pollutant accumulation that corresponds to the social optimum are summarized in the following proposition.

Proposition 2.2

(i) A unique optimal long run equilibrium (steady state) for the pollutant accumulation and its social cost defined as $(x^{*\infty}, \lambda^{*\infty}) : \dot{x} = 0, \dot{\lambda} = 0$, exists.

(ii) The steady state is a local saddle point.

(iii) For a small discount rate, the steady state is globally asypmtotically stable for bounded solutions of (9.1), (9,2).

Proof

(i) The isocines for the MHDS are defined for $\dot{x} = \dot{\lambda} = 0$. Solving (9.1) for x and substituting in (9.2), we obtain

$$g(\lambda)=(\rho+b)\lambda+D'\left(\frac{1}{b}\Sigma_i e_i(\lambda)\right)$$

$g(\lambda)$ is continuous with g(0)>0. Also, since D'>0, $e'_i>0$ and $\lambda<0$, there exists a λ^+ sufficiently large in absolute value such that e_i and D' are sufficiently small, so that $g(\lambda^+)<0$. continuity of g implies a value λ^*_∞ : $\lambda^+<\lambda^*_\infty<0$ such that $g(\lambda^*_\infty)=0$. By the monotonicity of g, λ^*_∞ is unique. Then the unique equilibrium value for the pollutant accumulation is

$$x^*_\infty=\frac{1}{b}\Sigma_i e_i(\lambda^*_\infty)$$

(ii) The Jacobian of the MHDS around the equilibrium point is

$$J=\begin{vmatrix}-b & \Sigma_i e'_i \\ D'' & (\rho+b)\end{vmatrix}$$

since $|J|<0$ the equilibrium point is a local saddle point (e.g. , Beavis and Dobbs 1990). This means that there exists a one dimensional manifold M that contains the equilibrium point. For initial conditions $(x_0,\lambda_0)\in M$ in the neighborhood of the equilibrium point it holds for the optimal solution that $(x(t)),\lambda(t))$ tends to the equilibrium point as $t\to\infty$.

(iii) Let $K\subset X\times R$ be a compact set. A solution of the MHDS will be bounded if $(x(t),\lambda(t))\in K$ for all $t\in[0,\infty)$. Form the curvature matrix

$$Q=\begin{bmatrix}-H_{xx} & \frac{\rho}{2} \\ \frac{\rho}{2} & H_{\lambda\lambda}\end{bmatrix}=\begin{bmatrix}D'' & \frac{\rho}{2} \\ \frac{\rho}{2} & \Sigma_i e'_i\end{bmatrix},\quad H_{xx}=\frac{\partial^2 H}{\partial x^2}\quad,\quad H_{\lambda\lambda}=\frac{\partial^2 H}{\partial\lambda^2}$$

This matrix is positive definite for small ρ. It follows from Brock and Scheinkman (1976) that the steady state is globally asymptotically stable for all bounded solutions, That is, for any initial condition on the stable manifold, the solution will converge to the steady state as $t \to \infty$. ■

The solution is illustrated in Figure 1.

Proposition 2.3 The optimal discharge levels for a profit maximizing firm, $\hat{e}_i$, will exceed the socially desirable levels for the same firm, e_i^*, i=1,...,n. As a result the corresponding equilibrium accumulation of the pollutant x_∞ will exceed the socially desirable level x^*_∞.

Proof $\hat{e}_i$ is defined as the solution of $B_i'(\hat{e}_i)=\lambda$ with $\lambda=0$ while e_i^* is defined as the solution of $B_i'(e_i^*)=\lambda$ with $\lambda<0$. Since $e_i'(\lambda)>0$, it follows that $\hat{e}_i>e_i^*$ all t. This implies that for all t, $x(t)>x^*(t)$ and in equilibrium $x_\infty>x^*_\infty$. This can also be seen from Figure 1 where x_∞ corresponds to the equilibrium accumulation for $\lambda=0$. ■

This proposition implies that if individual discharges were observable, an effluent tax set as $\tau(t)=-\lambda(t)$ would secure socially optimal discharge and accumulation levels. If, however, perfect monitoring is not possible, moral hazard would appear. To demonstrate that, let $\xi_i \in [0,1]$ be the probability that firm i 's emissions will be observed. For $\xi_i=1$ there is perfect monitoring while for $\xi_i=0$ the firm's emissions cannot be observed at all. For $\xi_i \in (0,1)$ there is incomplete monitoring that can be associated, for example, with random measurement of firms' emissions. The profit maximization problem for firm i becomes

$$\max_{e_i}[(1-\xi_i)B_i(e_i)+\xi_i[B_i(e_i)+\lambda e_i]],\ i=1,\ldots,n,\quad t\in[0,\infty)$$

The first order conditions for interior solution are

$$B_i'(e_i^+)=-\xi_i\lambda$$

It is clear that only in the case of perfect monitoring, that is $\xi_i=1$ for all i , $e_i^+=e_i^*$ and the socially desirable pollutant accumulate on level can be obtained through an effluent tax. Under incomplete monitoring, $\xi_i\in[0,1)$ and $e_i^+>e_i^*$ with $e_i^+=\hat{e}_i$ in the extreme case of $\xi_i=0$. Since $B_i(e_i^+)>B_i(e_i^*)$ the individual firms will not adopt the socially optimal discharge policy because by not doing so they can increase their expected profits. In terms of Figure 1, this implies that for $\xi\in(0,1)$, the long run pollutant accumulation level will be between $x^{*}{}_\infty$ and x_∞.

(ii) Uncertainty, $\sigma>0$

Applying the stochastic maximum principle (Malliaris and Brock 1982) , the generalized current value Hamiltonian is

$$H=\Sigma_i B_i(e_i)-D(x)+\lambda(\Sigma_i e_i-bx)+\frac{1}{2}(\sigma x)^2\lambda_x$$

where $\{\lambda(t)\}$ is a random process reflecting the social shadow cost of pollutant accumulation and

$$\lambda=\frac{\partial V(x)}{\partial x}=V_x\quad,\quad \lambda_x=\frac{\partial^2 V(x)}{\partial x^2}=V_{xx}$$

with V(x) being the optimal value function. This function solves the Hamilton-Bellman-Jacobi (H-B-J) equation

$$\rho V = \max_{e_i} \{ \Sigma_i B_i(e_i) - D(x) + V_x(\Sigma_i e_i - bx) + \frac{1}{2}(\sigma x)^2 V_{xx} \} \qquad (11)$$

Since the Hamiltonian is concave in (x, e_i), the H-B-J is concave in x, thus $V_{xx}<0$, reflecting society's risk aversion. Assuming in the following a quadratic optimal value function, V_{xx} is independent of x.

The optimality conditions can be written as

$$B_i'(e_i^*) \leq -\lambda \quad , \quad e_i^* \geq 0 \quad all \quad i \; , \quad t \qquad (12.1)$$

$$d\lambda = [(\rho+b)\lambda + D'(x) - \sigma^2 x \lambda_x] dt + (\sigma x)\lambda_x dz \qquad (12.2)$$

along with equation (2), and a transversality condition (Brock and Magill 1979) that holds for the optimal random processes x-(t) and $\lambda^*(t)$ which solve problem (P.1).

$$\sup E_o[\exp(-\rho t) \quad \lambda(t) x^*(t)] \leq 0 \quad as \quad t \to \infty \qquad (12.3)$$

From (12.1) the optimal discharge for interior solutions is determined as $e_i^* = e_i(\lambda)$. By substituting e_i^* into (12.2) and (2), a stochastic MHDS can be obtained. fissuring that a unique solution exists, this solution will be a diffusion process, with drift coefficients

$$[(\rho+b)\lambda + D'(x) - \sigma^2 x \lambda_x] \quad , \quad [\Sigma_i e_i(\lambda) - bx]$$

respectively and diffusion coefficients $(\sigma x \lambda_x)^2$ and $(\sigma x)^2$ respectively.

Some steady state properties. of the stochastic version of the MHDS defined above can also be presented. Assume the following:

(A.1) There exists a compact convex set $K \subset X \times R$ such that for all stochastic processes $x(t)$, $\lambda(t)$ with initial non-random conditions x_0, λ_0 that are solutions of the MHDS, it holds $(x(t), \lambda(t)) \in K$ for all $t \in [0, \infty)$.

(A.2) The optimal value function V(x) is strictly concave and differentiable.

(A.3) The transversality condition (12.3) is satisfied.

Proposition 2.4 Under assumptions (A.1) - (A.3) and for small ρ it holds that:

(i) All bounded solutions $(x(t), \lambda(t)) \in K$ converge almost surely as $t \to \infty$ to the optimal processes $(x^*(t), a^*(t))$, that are solutions of the problem (P.1) for any non-random initial condition $(x_0, \lambda_0) \in K$.

(ii) There exists a distribution function F(x) for the pollutant accumulation such that the distribution functions $F_t(x), t \in [0, \infty)$ converge to F(x) as $t \to \infty$. F(x) is the steady state distribution.

Proof

(i) Define the generalized curvature matrix

$$Q = \begin{bmatrix} -H_{xx} & \frac{\rho}{2} \\ \frac{\rho}{2} & H_{\lambda\lambda} \end{bmatrix} = \begin{bmatrix} D'' & \frac{\rho}{2} \\ \frac{\rho}{2} & \Sigma_i e_i' \end{bmatrix} \quad , \quad Q_{\lambda_x \lambda_x} = 0$$

Q is positive definite for small ρ, while the second derivative with respect to λ_x is non-negative. It follows, then from Brock and Magill (1979) that $x(t) \to x^*(t)$, $\lambda(t) \to \lambda^*(t)$ as $t \to \infty$ for any non-random initial condition.

(ii) Let $e_i^*(x) = e_i(x, V_x(x), V_{xx}(x))$ be the optimal policy function where V(x) solves the H-B-J equation and also let $f^*(x)=\sum e_i^*(x)-b(x)$, $\sigma^*(x)=\sigma x$. By proposition (i) above, $\|x(t)-x^*(t)\|\to 0$ in probability as $t\to\infty$, if also f*(x) satisfies a Lipshitz condition then following Brock and Magill (1979), there exists a stationary distribution for the pollutant accumulation. ■

Proposition 2.5 Let $(x_m^*)_\infty$ denote the expected long run equilibrium concentration of the pollutants then $(x_m^*)_\infty < x^*_\infty$.

Proof Take expected values in the integral form of the equations of the stochastic MHDS and denote $E(x)=x_m$ and $E(\lambda)=\lambda_m$. The following system of differential equations can be obtained for the expected values

$$\dot{\lambda}_m=(\rho+b)\lambda_m+E[D'(x)]-\sigma^2 x_m\lambda_x \qquad (13.1)$$

$$\dot{x}_m=\Sigma_i e_i(\lambda_m)-bx_m \qquad (13.2)$$

where $e_i(\lambda_m)$ is obtained by taking expected values in (12.1) and then solving for e_i^*.

In long run equilibrium, $\dot{x}_m=\dot{\lambda}_m=0^{(5)}$, then from (13.1) we obtain

$$\lambda_m=\frac{-E[D'(x)]+\frac{1}{2}\sigma^2 x_m\lambda_x}{(\rho+b)}$$

By Jensen's inequality $E[D'(x)]\geq D'(x_m)$, also $\lambda_x<0$. Therefore the isocine corresponding to $\dot{\lambda}_m=0$, lies everywhere below the isocine corresponding to $\dot{\lambda}=0$ for the deterministic case, and is relatively steeper (Fig. 1). This implies that $(x_m^*)_\infty < x^*_\infty$. ■

Thus society's risk aversion as reflected by the risk premium $-\sigma^2 x\lambda_x>0$ causes expected socially optimal pollutant concentration in the long run to be lower than the corresponding level for the deterministic case. The stability properties of system (13) are the same as those of the deterministic case.

Proposition 2.6 The expected long run pollutant concentration $(x_m)_\infty$ under profit maximization exceeds the corresponding socially desirable pollutant concentration $(x_m^*)_\infty$.

Proof This follows directly from propositions (2.1), (2.3) and (2.5). By these propositions, the following relation holds

$$(x_m)_\infty = x_\infty > x^*_\infty > (x_m^*)_\infty$$

The result is also illustrated in Figure 1. ∎

By the same proposition it follows that the deviation between profit maximizing and socially optimal concentration levels is greater under uncertainty. Thus although moral hazard can appear in a deterministic model in the absence of monitoring, the presence of uncertainty intensifies the problem in the sense that the gap between socially desirable and actual (profit maximizing) levels is greater relative to the deterministic case.

3. Intertemporal incentives

3.1 Efficient incentives and strategy spaces

In the absence of monitoring of individual discharges, the social planner can observe only deviations between desired and actual pollutant concentrations at some receptor point. The objective is to introduce an incentive scheme such that individual dischargers will be induced to follow a policy leading to a long run socially desirable level of pollutant accumulation. An incentive scheme with this property should depend on deviations between observed and desired levels at each instant of time. If deviations are observed, every potential discharger pays a penalty. If no deviations are observed, then no penalties are imposed. Deviation dependence is desirable since it can provide a basis for practical implementation. Furthermore, in the incentive scheme, deviations should be valued according to the social valuation of the pollutant accumulation. A scheme achieving the social planner's objective will be called efficient.

Condition for efficient incentive schemes Let $x(t)-x^*(t)$ be the deviation between observed and desirable pollutant concentration levels, with $x^*\infty$ being the socially optimal equilibrium level as $t\rightarrow\infty$. Let $\phi\equiv\phi(x(t)-x^*(t))$ be a function such as

$$\phi \gtreqless 0 \text{ as } x(t)-x^*(t) \gtreqless 0 \text{ with } \phi' > 0$$

Let $x(\phi,t)$ be a pollutant accumulation path resulting when profit maximizer dischargers are subjected to the incentive scheme ϕ. The incentive scheme ϕ will be efficient if $x(\phi,t)\rightarrow x^*\infty$ as $t\rightarrow\infty$.

The requirement that the incentive scheme be such that it results in convergence to the equilibrium point seems reasonable, since immediate adjustment to the optimal path may require undesirable production cuts.

The analysis of incentive schemes is carried out in the context of an n-player non-cooperative dynamic game. The choice of the strategy space for these differential games depends on information structures (e.g., Basar and Olsder 1982, Fershtman and Kamien 1915). In the following, the analysis is limited to the often-employed open-loop and feedback structures.

Discharger i's information structure is said to be

(S1) Open-loop (OL) if $e_i = \theta_i(x_0, t)$ $\quad i=1,\ldots,n$

(S2) Feedback (FB) if $e_i = \theta_i(x, t)$ $\quad i=1,\ldots,n$

An (OL) or (FB) strategy is a time path $\{e_i(t)\}$ such that $e_i \in E_i$, all i and t.

Open-loop Nash equilibrium (OLNE) and feedback Nash equilibrium (FBNE) are defined for the strategy spaces [6] corresponding to the OL and FB information structures. OLNE solutions correspond to an infinite period of commitment. Players, that is dischargers, commit themselves to a particular path at the outset of the game and do not respond to observed variations of the pollutant concentrating. Discharge paths that constitute equilibrium for the game that starts at (x_0, t_0) do not constitute equilibrium for the game that starts at a different (x_0, t_0). Thus OLNE is not subgame perfect and this implies time inconsistency, a discharge policy that is optimal at the outset of the game is not optimal at a later period. Feedback strategies, on the other hand, depend on current ambient concentration levels. Firms do not commit themselves at the outset of the game and the FBNE is an equilibrium for any initial condition, thus it constitutes a subgame perfect (Selten 1975, Fershtman 1987,1988, Reiganum and Stokey 1985). The feedback equilibrium can be generalized to account for the conjectures of discharger i about the discharges of the rest of the firms. In this case, the strategy of the ith player is defined as

(S3) Feedback Complete Conjecture (FBC): $e_i = \theta_i(x, \underline{e}_{-i}, t)$ where $\underline{e}_{-i} = (e_1, \ldots, e_{i-1}, e_{i+1}, \ldots, e_n)$. The FBCNE is a subgame perfect (Fershtman and Kamien 1985).

The payoff of firm i under the incentive scheme ϕ is defined as

$$J^i(e_1^*, \ldots, e_n^*) = E_0 \int_0^\infty \exp - \rho t [B_i(e_i) - \phi(x - x^*)] dt \quad i = 1, \ldots, n$$

each firm tries to maximize its payoff subject to (2) with $e_i \in E_i$. The OLNE (or FBNE or FBCNE) equilibrium is defined as an n-tuple of OL (or FB or FBC) strategies $(e_1^*, \ldots, e_n^*)$ where e_i is defined in (S1) - (S3) such that

$$J^i(e_1^*, \ldots, e_n^*) \geq J^i(e_1^*, \ldots, e_{i-1}^*, e_i, e_{i+1}^*, \ldots, e_n^*) \quad all\ i$$

In the following we examine the structure of the intertemporal efficient schemes that correspond to strategies (S1) - (S3).

3.1 Incentives under certainty

Efficient schemes corresponding to OLNE, FBNE and FBCNE are examined under the assumption that $\sigma = 0$.

(i) OLNE

Proposition 3.1 Let $\lambda^*(t) < 0$ be the social shadow cost of pollutant accumulation as defined by the solution of (P.1), then $\phi(x - x^*) = -\lambda^*(\rho + b)(x - x^*)$ is an efficient incentive scheme for OLNE.

Proof The current value Hamiltonian for the ith firm is defined as

$$H^i = B_i(e_i) + \lambda^*(\rho + b)(x - x^*) + \mu_i(e_i + \sum_{j \neq i} e_j^* - bx) \quad (14)$$

where $\underline{e}^*_{-i}$ is the vector of the optimal responses of the rest of the firms. The necessary conditions for optimality are

$$B_i'(e_i^*) \leq -\mu_i, \quad e_i^* \geq 0 \quad all\ i \tag{15.1}$$
$$\dot{\mu}_i = (\rho + b)(\mu_i - \lambda^*) \quad all\ i \tag{15.2}$$
$$\dot{x} = \Sigma_i e_i(\mu_i) - bx \tag{15.3}$$

Since ρ, b, λ are common for all i, it follows that $\mu_i = \mu, \forall i$. From (15.1) we obtain in equilibrium $(\dot{\mu}=0)$, that $\mu_\infty = \lambda^*_\infty$. Denote with x_∞^{OL} the equilibrium pollutant concentration under OLNE. From (15.3) we obtain

$$x(\phi, t) = x_\infty^{OL} \to \Sigma_i \frac{e_i(\lambda_\infty^*)}{b} = x_\infty^* \quad as\ t \to \infty \qquad \blacksquare$$

The result is illustrated in Figure 2.

The above incentive scheme is a type of effluent tax per unit of observable deviation between measured and desired accumulation levels. Under this scheme, once deviations are detected, every firm pays the same total amount[7], in contrast to the standard Pigouvian taxes where the total amount paid depends on individual discharges. It should also be noted that if past overdischarges caused deviation from the optimal path, then firms will pay the charge during the period of the adjustment to the optimal path, even if they currently follow optimal discharge policies.

The existence of a unique equilibrium point can be established by an argument similar to the one used in proposition (2.2.i). It is necessary, however, to examine the stability properties of the model in order to be sure that the proposed incentive scheme does not result in completely unstable equilibria.

<u>Proposition 3.2</u> The steady state solution $(x_\infty^{DL}, \mu_\infty)$ is

(i) A local saddle point.

(ii) Globally asymptotically stable for bounded, solutions.

<u>Proof</u>

(i) The Jacobian of the MHDS (15.3), (15.2) is defined as

$$J=\begin{vmatrix} -b & \sum_i e_i' \\ 0 & (\rho+b) \end{vmatrix}$$

since $|J|<0$, the equilibrium point has the saddle point property.

(ii) Following Sorger (1989), the curvature matrix can be written as (8)

$$K^\gamma=\begin{bmatrix} H_{xx}+\gamma\,[H_{x\mu}+H_{\mu x}-\rho]+\gamma^2 H_{\mu\mu} & -\frac{\rho}{2} \\ -\frac{\rho}{2} & -H_{\mu\mu} \end{bmatrix}=\begin{bmatrix} \gamma(-2b-\rho)+\gamma^2\Sigma_i e_i' & -\frac{\rho}{2} \\ -\frac{\rho}{2} & -\Sigma_i e_i' \end{bmatrix}$$

since $\mu(t)$ is bounded by the bounded solution assumption, if the e_i functions have bounded slope then there exists a finite $\gamma^+ \in (0,(2b+\rho)/\sum e_i')$. For $\gamma=\gamma^+$ and sufficiently small ρ, the curvature matrix is negative definite. Thus the steady state is globally asymptotically stable for bounded solutions. ∎

Solutions of the type described above suffer from the multiplicity of informationally non-unique Nash equilibria, A way of removing informational non-uniqueness is to restrict the equilibrium solution concept to a feedback Nash equilibrium (Basar and Olsder 1982). This type of restriction requires that players have access to the current value of the state. In the model described here, this is a plausible assumption since it can be assumed that the information that the social planner (environmental agency) has about the current accumulation of the pollutant becomes public knowledge without delay.

(ii) FBNE

In analyzing this type of equilibrium, the cross effects that describe conjectural variations make it very difficult if not impossible to study the problem in the general form. To obtain some insight into this type of equilibrium, a specific simple form for the conjecture function is assumed. In particular, for the conjecture function of firm i we assume

(AC1) $e_j = c_j + ax \;,\; a<0 \;,\; j \neq i \;,\quad i,j=1,\ldots,n$

This conjecture function indicates that firm i expects other firms' discharge functions to contain an autonomous part and a part that depends linearly on current ambient concentration levels.

Proposition 3.2 Under (AC1) the efficient incentive scheme for FBNE takes the form

$$\phi(x-x^*) = -\lambda^* [(\rho+b)-(n-1)a](x-x^*) \quad (9)$$

Proof The current value Hamiltonian for this problem is

$$H^i = B_i(e_i) - \phi(x-x^*) + \mu_i [e_i + \sum_{j \neq i} (c_j + ax) - bx] \quad (16)$$

The optimality conditions are (15.1), (15.3) and

$$\dot{\mu}_i = [(\rho+b)-(n-1)a](\mu_i - \lambda^*) \quad i=1,\ldots,n \quad (17)$$

Thus $\mu_i=\mu$ all i. In equlilibrium $\mu_\infty=\lambda^*_\infty$, and

$$x(\phi,t)=x_\infty^{FB}-\sum_i \frac{e_i(\lambda^*_\infty)}{b}=x^*_\infty \quad as\ t\to\infty \qquad \blacksquare$$

The stability properties of the steady state can be analyzed in the same way as with OLNE. The equilibrium point is a local saddle point and globally asymptotically stable for bounded solutions.

(iii) FBCNE

To analyze this type of equilibrium, the following conjecture function for firm i is postulated

$$(AC2)\quad e_j=c_j+ax+\beta\sum_{k\neq j} e_k,\ j\neq i,\ i,j=1,\ldots,n,\ k=1,\ldots,j-1,j+1,\ldots,n \quad (10)$$

The third term of (AC2) reflects the fact that firm i expects firm j to adjust, its discharges by taking into account the discharges of all other firms. If we assume that all i=1,...,n firms are similar, it is not unreasonable to expect that the same weight be given to each of the rest if the firms' discharges, by every firm.

Proposition 3.4 Under (AC2) the efficient scheme for FBCNE takes the form

$$\phi(x-x^*)=-[\lambda^*-(n-1)\beta]\,[(\rho+b)-(n-1)\alpha]\,(x-x^*)$$

Proof The current value Hamiltonian for the problem is

$$H^i=B_i(e_i)-\phi(x-x^*)+\mu_i[e_i+\sum_{j\neq i}(e_j+ax+\beta\sum_{k\neq j}e_k)-bx] \quad (18)$$

The optimality conditions are

$$B_i'(e_i^*) \leq -z_i \quad , \quad z_i = \mu_i + (n-1)\beta \quad , \quad e_i^* \geq 0 \qquad (19.1)$$
$$\dot{\mu}_i = [(\rho+b)-(n-1)a]\mu_i - [\lambda^* - (n-1)\beta][(\rho+b)-(n-1)a] \qquad (19.2)$$
$$\dot{x} = \Sigma_i e_i(z_i) - bx \qquad (19.3)$$

$\mu_i = \mu$ all i, thus in equilibrium we have $\mu_\infty = \lambda^*_\infty - (n-1)\beta$. Substituting into (19.3), we obtain

$$x(\phi, t) = x_\infty^{FBC} \to \frac{\Sigma_i e_i(\lambda_\infty^*)}{b} = x_\infty^* \quad as \quad t \to \infty \qquad \blacksquare$$

The stability properties of the steady state are similar to OL and FB solutions.

To compare the three incentive schemes derived above, let

$$\tau_1 = -\lambda^*(\rho+b)\,,\ \tau_2 = \tau_1 + \lambda^*(n-1)a,\ \tau_3 = \tau_2 + (n-1)\beta[(\rho+b)-(n-1)a]$$

denote the taxes per unit of deviation under OL, FB and FBC respectively. For a<0, $\tau_2 > \tau_1$. If firm i expects other firms to reduce their emissions when concentration increases, it has incentive to overdischarge, thus a relatively higher tax is required. A similar result applies if $\beta > 0 (a < 0)$. Then, $\tau_3 > \tau_2 > \tau_1$. If, however, $\beta < 0$, some inequalities might be reversed.

3.1.1 Comparison with static incentive schemes

In a static context, that is when the dynamics of the pollutant accumulation are ignored, the efficient differentiable incentive scheme charges the full social cost of deviations between observed and desired ambient concentration levels (Segerson 1988, Xepapadeas 1991).

This can be easily demonstrated along the lines of the model developed so far. The social planner solves the problem

$$\max_{e_i, x} \Sigma_i B_i(e_i) - D(x)$$
$$s.t. \quad x = \Sigma_i e_i$$

The optimality conditions imply

$$B_i'(e^*) = -\lambda^* = D'(x^*)$$

thus $\lambda < 0$ is the marginal social cost of pollutant concentration. Discharger i will follow a socially desirable policy if he faces the incentive scheme $\lambda^*(x-x^*)$. This is a standard non-balanced budgeting contract for preventing moral hazard in teams (Holmstrom 1982) . Under this scheme, discharger i's problem is

$$\max_{e_i} B_i(e_i) + \lambda^*[(e_i + \sum_{j \neq i} e_j^*) - x^*]$$

where e^*_{-i} is the optimal response vector. Nash equilibrium implies $B_i'(e_i^*) = -\lambda^*$ which is the condition for the static social optimum. We are in a position now to analyze the implications. for the long run pollutant accumulation from the adoption of static incentive schemes.

<u>Proposition 3.5</u> Under static differentiable incentive schemes of non-budget balancing type($\lambda^*(x-x^*)$), the long run pollutant concentration level compares as follows to the socially desirable level

$$(i)\, x_\infty^{OL} < x_\infty^*, \quad (ii)\, x_\infty^{FB} > x_\infty^{OL} \lesseqgtr x_\infty^*, \quad (iii)\, x_\infty^{FBC} \lesseqgtr x_\infty^*$$

Proof

(i) Under OLNE the following differential equation is satisfied for the shadow cost of the pollutant

$$\dot{\mu}=(\rho+b)\mu-\lambda^{*}$$
$$\text{or at equilibrium } \mu_{\infty}^{OL}=\frac{\lambda_{\infty}^{*}}{\rho+b}<\lambda_{\infty}^{*}$$

under the plausible assumption that $0<\rho+b<1$, (where $\lambda,\mu<0$). This implies that

$$e_i(\mu_{\infty}^{OL})<e_i(\lambda_{\infty}^{*})\,\forall i \quad , \quad x_{\infty}^{OL}<x_{\infty}^{*}$$

The result is illustrated in Figure 3.

(ii) Under FBNE we have

$$\dot{\mu}=[\rho+b-(n-1)a]\mu-\lambda^{*}$$

or in equilibrium

$$\mu_{\infty}^{FB}=\frac{\lambda_{\infty}^{*}}{\rho+b-(n-1)a}>\mu_{\infty}^{OL}\lesseqgtr\lambda_{\infty}^{*} \quad \text{as} \quad [(\rho+b)-(n-1)a]\lesseqgtr 1 \;, \quad (a<0) \quad \text{thus}$$
$$e_i(\mu_{\infty}^{FB})>e_i(\mu_{\infty}^{OL})\lesseqgtr e_i(\lambda_{\infty}^{*}) \;, \quad x_{\infty}^{FB}>x_{\infty}^{OL}\lesseqgtr x_{\infty}^{*} \quad (Fig.\ 3)$$

(iii) Under FBC we obtain in the same way

$$e_i^{*}=e_i(z_{\infty}^{*}) \text{ where } z_{\infty}^{*}=\mu_{\infty}^{FBC}+(n-1)\beta=\frac{\lambda_{\infty}^{*}}{(\rho+b)-(n-1)a}+(n-1)\beta\lesseqgtr\lambda_{\infty}^{*} \quad \text{thus}$$
$$x_{\infty}^{FBC}\lesseqgtr x_{\infty}^{*} \quad (Fig.\ 3) \qquad \blacksquare$$

The above results imply that “static” incentive schemes that charge the full cost of observed deviation between measured and desired pollutant concentration levels, when pollutant accumulation is a dynamic process, lead in general to supoptimal results. Under OL strategies, the “static” incentives lead to overabatement since their corresponding charge per unit deviation is greater than the charge required for long run convergence to the desired levels. Under more complicated strategies, however, the optimal dynamic charge per unit deviation could be higher than the static one, and in this case the static scheme would result in underabatement.

3.2 Incentives under uncertainty

We proceed to examine the structure of an efficient incentive scheme in the stochastic framework defined in (2) by analyzing the FBNE.[11] In the stochastic framework efficiency is defined in terms of equilibrium expected value of the pollutant concentration under the scheme. That is, the incentive scheme is efficient if

$$E[x(\phi,t)] \to E(x_m^*) = (x_m^*)_\infty \quad as \quad t \to \infty$$

with $(x_m^*)_\infty$ as defined in proposition 2.5.

<u>Proposition 3.6</u> Under feedback strategies with conjecture functions as defined in (AC2), the efficient incentive scheme for the stochastic framework takes the form

$$\phi_i(x-x_m^*) = -\{[\lambda_m^*[(\rho+b)-(n-1)a] - \frac{1}{2}(\sigma^2 x)\mu_x^i](x-x^*) - \frac{1}{2}(\sigma^2 x)\mu_x^i x^*\}$$
$$where \ x_m^* = E[x^*(t)] \ , \ \lambda_m^* = E[\lambda^*(t)]$$

as defined in proposition 2.5 and $\mu_x^i = V_{xx}^i$ with $V'(x)$ being the optimal value function that solves the H-B-J equation for the ith firm.

<u>Proof</u> The generalized current value Hamiltonian is defined as

$$H_i = B_i(e_i) - \phi^i(x-x^*) + \mu_i[e_i + (n-1)ax - bx] + \frac{1}{2}(\sigma x)^2 \mu_x^i$$

The optimality conditions can be written as

$$B_i'(e_i^*) \leq -\mu_i \qquad (20.1)$$
$$d\mu_i = [(\rho+b) - (n-1)a](\mu_i - \lambda_m^*)]dt + (\sigma x)\mu_x^i dz \qquad (20.2)$$
$$dx + [\Sigma_i e_i(\mu_i) - bx]dt + (\sigma x)dz \qquad (20.3)$$

Taking expected values for (20.2),(20.3) in integral form, we obtain the following system of differential equations for x_m, λ_m

$$\mu_m = [(\rho+b) - (n-1)](\mu_m - \lambda_m^*) \quad since\ \mu_m^i = \mu_m\ all\ i \qquad (21.1)$$
$$x_m = \Sigma_i e_i(\mu_m) - bx_m \qquad (21.2)$$

It can be easily seen that in equilibrium

$$E[x(\phi,t)] \to (x_m^*)_- \quad as \quad t \to \infty \qquad \blacksquare$$

The equilibrium point for system (21) is a local saddle point and globally asymptotically stable for bounded solutions, as can be easily shown by following the approach in proposition 2.2.

Some observations are in order with respect to the scheme of proposition 3.6. The deviations from the expected optimal path are valued according to their expected social cost and according to a risk premium that reflects discharger i's risk aversion. Thus the charge per unit deviation from the expected path is discriminatory. A uniform tax can be obtained under an assumption of “equal curvature” for optimal value functions $V^i_{xx} = V_{xx}$ all i.

The proposed incentive scheme might result in subsidies or penalties even if all firms follow desirable discharge policies because of random fluctuations that cause deviations from the expected path. One way of improving the scheme so that it does not result in a case where there is a continuous switch from subsidies to penalties and vice versa is to supplement it with a type of “confidence belt”. Observed values outside this belt would not be regarded as resulting from random fluctuations and the charges would be imposed. One way of defining the belt is to use the diffusion property of the solutions of system (12.2), (2). By this property, a limit $x^+(t')$ can be defined such that the probability of the pollutant accumulate on exceeding this limit when all dischargers follow optimal policies is less than a predetermined probability, $Pr(x(t') \geq x^+(t')/e_i^*) \leq w$. If in time t', $x(t') > x^+(t')$, this can be regarded as establishing overemissions “beyond any reasonable doubt”. Then the charge per unit deviation is triggered. Of course, if the excess accumulation of the pollutant is a result of events which are clearly beyond the control of the specified set of discharging firms (e.g., an environmental disaster caused by a third party), charges are not imposed. On the other hand, subsidies could never be paid. It is clear that they are the result of random factors since there is neither the incentive nor is it desirable to emit below the socially optimal levels.

4. Concluding remarks

Incentive schemes developed so far to deal with non point-source pollution problems are essentially static since they ignore the implication of the dynamics of pollutant accumulation, An attempt is made in this paper to develop incentive schemes which account for the dynamics of environmental change.

When individual emissions are not observable and consequently Pigouvian taxes are ineffective, profit maximizing firms emit more than is socially desirable. As a result, pollution accumulation levels exceed the respective socially optimal levels.

It has been demonstrated that incentive schemes can be constructed such that the path of pollutant accumulation under the scheme converges to the equilibrium socially desirable pollutant accumulation level. The schemes take the form of charges per unit of observed deviation between measured and desired levels. The charge depends on the pollutant's shadow cost, on the discount rate, on the natural pollution decay rate and on parameters associated with the information structure of the model. When discharging firms follow feedback strategies, the charge is higher as compared to the open-loop case. In general., it is expected that firms will follow feedback strategies, since these types of strategies do not imply long period% of commitment and public information about the state of the pollutant accumulation makes their employment feasible. Similar incentive schemes can be constructed for a model with environmental uncertainty. The charge per unit deviation depends additionally on risk premiums under the appropriate risk aversion assumptions.

In general, application of static incentive schemes in dynamic situations results in suboptimalities, In particular, if dischargers adjust their emission policy according to current pollution accumulation (feedback strategies), application of static schemes may result in over accumulation of the pollutant in the long run.

Successful application of the incentive schemes requires the determination of the optimal path for pollutant accumulation. This could be a formidable task since it requires information on firms’ production and abatement technologies, damages from pollutant accumulation, the natural characteristics of pollutant decay, and the information structure used by the discharging firms. On the other hand, the proposed scheme once approximated provides a flexible mechanism for dealing with dynamic non point-source pollution problems, since it can be treated as a simple Pigouvian tax on deviations from the optimal path. Incentives will, however, be ineffective if there are dischargers operating outside the incentive scheme and their contribution to the accumulating of the pollutant cannot be distinguished from the contribution of the dischargers that are subjected to the scheme.

FOOTNOTES

1. The benefit function is defined as

$$B_i(e_i) = \max_{q_i} \Pi(q_i, e_i) = \max[pq_i - C_i(q, e_i)] \quad all\ t$$

where q_i is the firm's output, p is the output price (reflecting marginal utility) and $C_i(.,.)$ is a strictly convex cost function. Since $\Pi(q_i, e_i)$ is concave in q_i, e_i, max $\Pi(q_i, e_i)$ is concave in e_i (Kamien and Schwartz 1981). We furthermore assume throughout the paper that private and social benefits and costs coincide.

2. The stochastic processes $\{x(t)\}$ and {z(t)} are defined as $x(t,\omega)$, $z(t,\omega)$ where $\omega \in \Omega$, $t \in [0,\infty)$ and $(\Omega, \mathcal{F}, P)$ is a complete probability space with $\mathcal{F}$ being a o-field on Ω and P a probability measure on $\mathcal{F}$. In the text, ω and in most cases t are suppressed.

3. A unique solution exists because the coefficient functions of (2) satisfy the Lipschitz and growth conditions and the initial condition is non-random (Gard 1988). Furthermore the solution is positive for the positive initial condition (Chang and Malliaris 1987).

4. A steady state probability distribution for the pollutant accumulation exists with the steady state density function satisfying (Malliaris and Brock 1982, Merton 1975):

$$\pi(x) = \frac{m}{(\sigma x)^2} \exp\{\int^x \frac{2(\Sigma_i \hat{e}_i - by)}{(\sigma y)^2} dy\}$$

$$with\ m: \int_0^\infty \pi(x)\, dx = 1$$

The existence of this distribution can be proven by showing that 0 and ∞ are repelling boundaries. This implies that the solution will neither explode or degenerate to zero (for this approach, see Gard 1986).

5. The existence of a unique steady-state can be shown in a similar way as is shown in proposition 2.2 by noting that by Jensen's inequality $E[D'(x)]\geq D'(x_m)>0$.

6. The strategy space is the set of all possible OL or FB strategies.

7. Since at this stage the model is deterministic and since firms have incentive to discharge more rather than less, it must hold that $x(t)\geq x^*(t)$, all t. Thus under certainty the scheme always works as a tax and not as a subsidy.

8. This approach is used because the maximized Hamiltonian is linear in x, that is $H_{xx}=0$.

9. The structure of the incentive scheme depends on the conjecture function. For example, if

$$e_j=c_j+\frac{1}{2}ax^2 \quad then$$
$$\phi=-\lambda^*[(\rho+b)(x-x^*)-(n-1)\frac{a}{2}(x-x^*)^2]$$

10. For $\beta=0$ we are in FB strategies while for $a=\beta=0$ we are in OL strategies.

11. Formulating the problem in the stochastic framework eliminates informational non-uniqueness. Furthermore it can be shown that if the players do not have access to the current state (as for example in OLNE), then general conditions for Nash equilibri-

um solutions under uncertainty cannot be obtained (Basar and Olsder 1982).

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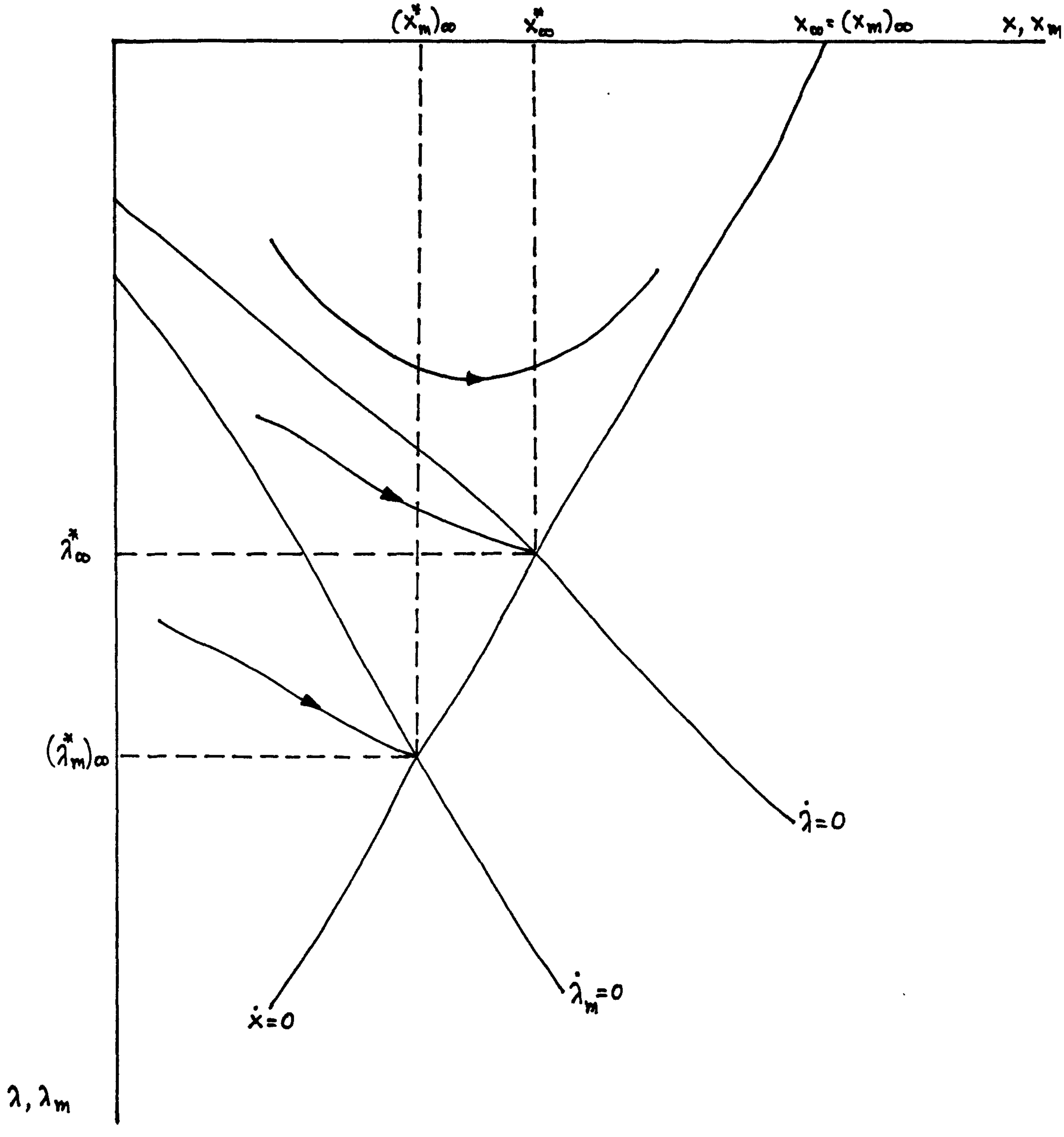

$(x_m^*)_\infty$
x_∞^*
$x_\infty = (x_m)_\infty$
x, x_m
λ_∞^*
$(\lambda_m^*)_\infty$
$\dot\lambda = 0$
$\dot\lambda_m = 0$
$\dot x = 0$
λ, λ_m

FIGURE 1

May 28, 1991

Differences in the Transaction Costs of Strategies to Control Agricultural Offsite and Undersite Damages

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Paper for the 1991 AERE Workshop on the Management of Nonpoint-Source Pollution, held June 6 and 7 in Lexington, Kentucky and hosted by the University of Kentucky

Differences in the Transaction Costs of Strategies to Control
Agricultural Offsite and Undersite Damages

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May 28, 1991

Differences in the Transaction Costs of Strategies to Control Agricultural Offsite and Undersite Damages

K William Easter*

Pollution of our water supplies by agricultural chemicals has been an area of growing concern since the second half of the 1980's when agricultural chemicals were found in many water samples from wells and springs across the United States. This was added to previous information that identified agricultural chemicals as an important source of surface water pollution. However, most efforts to alter agricultural chemical use have not been to prevent water pollution. Regulatory efforts have focused on preventing the hazardous health effects of pesticides during application and in keeping pesticide residues out of food.

Identifying and controlling the major sources of nonpoint agricultural chemical pollution are not easy. In most cases, farmers decide what, how much, and in what manner agricultural chemicals and animal waste" products will be applied to their lands. As a result they strongly influence how much may eventually reach surface or ground water supplies. Farmers' decisions are dictated by their own utility maximizing behavior and government policies and institutional arrangements that constrain or enhance their decision set (Figure 1). Soil type, topography, vegetation and climatic events all influence chemical movements towards various water sources as will farming practices. While farmers have little control over climatic events they can change farming practices and vegetative cover to alter the impacts of climatic events. Thus farmers' decisions and the policies and institutional arrangements that influence their decisions are critical in controlling agricultural chemical pollution from the use of fertilizers and pesticides.

When evaluating alternative strategies and policy instruments for controlling pollution, economists have focused on the efficient use of production resources and largely ignored transaction costs. They determine what tax or other policy instrument would be the least distorting in making producers internalize the externalities they create. However, the major costs involved in reducing water pollution in agriculture are likely to be the transaction costs of enacting and implementing alternative strategies and not distortions in production efficiency.

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This paper focuses on the differences in transaction costs of policies that change farmers' decisions concerning the use of chemicals and agricultural waste products (primarily manure). The effects of policy instruments and institutional arrangements are likely to be different on ground water than on surface water, suggesting that strategies for controlling agricultural chemical pollution must be carefully designed to account for differences in water sources as well as other physical and socioeconomic differences. One simple, nationwide strategy is not likely to be the most efficient in terms of either production or transaction costs.

Why Undersite and Offsite Damages?

Since many water sources polluted in rural areas are used by farmers, i.e., domestic wells, one may ask why farmers pollute their own water supply or that of their neighbors. There are, at least, five answers to this question. One is that farmers lack the knowledge or information concerning the adverse impacts that their farming practices and input uses have on water quality or more specifically, "their" water supply. A second explanation is that they are not concerned about water pollution costs imposed on their neighbors or those living downstream. This is the classic spatial and temporal, externality problem where upstream producers damage the water supply of downstream users, but not their own. Third, they may have decided that the use of chemicals or disposal of manure and the resulting increased income is more important than clean water. They may even be willing to buy bottled water instead of reducing chemical or manure applications. A fourth reason may involve imperfect information concerning the optimum use and application of inputs. For example, many livestock farmers in southeastern Minnesota apply 60 to 100 lbs. more nitrogen, in the form of manure, than is required for optimum crop production because of the lack of information concerning its nutrient value (Legg, 1991). The fifth reason is risk and uncertainty concerning economic and weather conditions that will affect crop production. Applying extra chemicals may help reduce weather related income losses. Thus, there is no one simple answer to the question. It is a combined problem of imperfect information externalities, risk, farmer income requirements and waste disposal.

An added reason for water pollution is the lack of clearly specified property rights concerning water quality for either surface water or ground water. Do consumers have the right to clean water, or do producers have the right to pollute the water? If the water is polluted, who has to pay to clean it up? In most cases, farmers are not prevented from polluting water supplies, and if a clean up is required, they generally do not pay any more than other consumers or taxpayers. Holding farmers financially liable for water pollution would clearly provide an incentive to stop water pollution and help internalize the externality.

Farmer Decisions

Farmers make long run capital decisions, such as the type of manure handling facility to install or farming system to use, that have important impacts on their chemical use and the transaction costs of changing chemical use. These decisions will depend on a number of uncertainties, including future commodity and chemical prices. Annual chemical use decisions are constrained by the capital assets that are in place. The farmer will decide on chemical use rates and timing based on crops selected, prices, weather, manure supplies, labor available, management capability and soil conditions. These decisions may change during the growing season in response to rainfall and temperature conditions. A heavy rainfall in areas with sandy soil may mean last week's fertilizer application has been lost and needs to be replaced. In contrast, dry conditions mean that less nitrogen is needed and different pest control practices may be required.

Management availability and risk play an important role in these short and long run decisions. Nitrogen in the U.S. is relatively cheap and pest control with herbicides and insecticides does not require as intense management as does mechanical and biological pest control. Furthermore, price and weather uncertainty along with the demands of part-time jobs encourage farmers to err on the high side of chemical use. A little extra nitrogen may increase crop yields in a good rainfall year by 10 to 20 percent. Also, if farmers do not control weeds early in the season with heavy use of herbicides, wet weather may prevent them from getting into their fields and applying the needed weed control until labor is scarce or the crop is too tall. Failure to control the weeds can result in as much as a 25 percent reduction in crop yields.

Differences between Surface and Ground Water

Externalities appear to be the most important explanation of surface water pollution since much of the damage occurs offsite or downstream. This explanation does not hold in all cases since local fish kills and lake pollution may directly impact the farmers that cause the pollution. Still, a major reason for surface water pollution is the external nature of the costs imposed by the pollution, while lack of information and income requirements are more important for ground water. Many externalities associated with ground water are localized while for surface water, they may occur in the next county or state.

The transaction costs of monitoring and enforcement would be quite different for surface and ground water. For surface water the problem is its mobility and the numerous sources of agricultural nonpoint pollution. Whose pesticides caused the fish kill? While it maybe difficult to identify the polluters of surface water, the most likely suspects are upstream farmers. Yet the mobility of surface water means that transaction costs of ex-post measure of contamination are likely to be high (frequent monitoring). For ground water pollution, monitoring and enforcement are also likely to be expensive because of the cost of monitoring sites. In many cases the monitoring of existing wells is not enough and special monitoring wells are necessary to locate contamination and polluters so that ground water quality standards can be enforced.[1]

[1] Pollution of ground water as well as surface water by pesticides appears to be highly related to improper use, storage or disposal of pesticides or extreme rainfall events following pesticide applications, with the exception of a few herbicides such as Atrazine. With extreme rainfall events, pesticide movement is generally accompanied by high levels of soil erosion, but not always. Ground water pollution appears to occur with normal application of nitrates and Atrazine, particularly on lighter soils. In the case of surface water, nitrate and Atrazine pollution is more closely related to high rainfall events.

Another important difference between surface and ground water pollution has to do with the values of water uses precluded by chemical pollution. There are, at least, two aspects to this difference. One is that surface water has a wider array of uses then does ground water. Irrigation, industrial, commercial and domestic water consumption are the main uses of ground water while surface water can also provide a long list of recreational opportunities. The second aspect is that the duration of pollution may be quite different between surface and ground water, particularly if the surface water is a stream or river. Many of the agricultural chemicals that contaminate water supplies are not as long lasting in the surface water as they are in the ground water. How these two aspects will influence the value of lost water uses will vary by location and water use. For example, when the ground water is, or might be, used for domestic consumption and no good alternative sources of water are available, the losses from pollution will be quite high. In contrast, if the ground water is used for irrigation and is not likely to be demanded for other uses, then the pollution losses are likely to be relatively small.

Benefits from Improved Water Quality

With both the amount of agricultural chemicals entering the water supplies and the demand for higher water quality increasing, the benefits from improving water quality are on the rise. The increased demand is due, in part, to the growth in U.S. incomes and population, as well as greater knowledge concerning the harmful nature of certain agricultural chemicals. The growth in demand for bottled water and water based recreation are both directly related to this increased demand for higher water quality. Of course, water for household uses requires a different level of water quality than does water for recreational uses. Yet agricultural chemicals have damaged water for both of these uses.

Recreational benefits are among the largest, if not the largest, class of potential benefits from surface water pollution control (Rogers, et. al., 1990). Currently, they exceed the health or other water treatment benefits from reduced surface water pollution. In contrast, the primary concern in ground water appears to be the potential health effects or the increased cost of water treatment. In a number of cases, chemical pollution of ground water has forced the closing of wells and caused shifts to alternative water sources.

On the supply side of pollution, there are certain geographic areas that are more susceptible to water pollution and, therefore, they offer higher returns from pollution control efforts. For ground water, these are likely to be areas with light soils and shallow aquifers, or karst aquifers. The susceptible areas are not as easy to identify for surface water. However, surface water sources surrounded by moderately or steeply sloping, intensively farmed lands are clearly susceptible to agricultural chemical pollution. Thus, the physical characteristics of land, climatic conditions, amounts and types of chemicals, and farming practices will all be important in determining the degree of chemical contamination and level of benefits from pollution control.

On the demand side of pollution abatement, growth in per capita income and in population, the availability of alternative water supplies and the cost of pollution cleanup will all be important. These factors help determine the value of protecting water quality for a range of water uses. Clearly, areas with large populations and low rainfall, such as Los Angeles, will have a high demand for good quality water and programs that prevent agricultural water pollution. However, given where Los Angeles must obtain much of its water supply, it has a very limited capacity to influence what agricultural chemicals get into “their” water supply. For example, water taken from the Colorado River to supply L.A. will contain agricultural chemicals that have come from farms as far away as Colorado and Wyoming. Thus the demand for public action or changes in property rights concerning water pollution from agricultural chemicals is growing in urban American and will continue to expand. In addition, because of water’s mobility the demand for clean water may come from areas outside the source of supply as is the case for LA.

The cost of cleaning up polluted ground water is sufficiently high, in a number of aquifers, to preclude it as an efficient alternative. In contrast, we have been cleaning up polluted rivers for many years at a wide range of costs. The persistence and toxicity of the pollutants are both important in determining the cost of clean up. Finally, the benefits from preventing water pollution will be closely related to the cost of substitute water supplies and the intended uses to which they will be devoted. To illustrate, if water is used for irrigation, there will be little or no loss from nitrate contamination, but the losses could be substantial if the use shifts to human consumption. The demand for “cleaner” water will also depend on whether the pollutant causes cancer or just tastes bad during a few weeks in the spring. When the demand is for domestic water use and the clean up costs are high, with no good substitute supplies available, then the benefits from protecting the water source from agriculture pollution will be high, especially if the water source is susceptible to contamination (Figure 2).

Pollution control policies need to be directed at those areas and types of water uses where the highest net benefits to society can be achieved from protecting the water supplies. In addition, policies, programs and institutional arrangements need to be designed so that the cost of such protection is minimized. One of the critical costs that should be minimized is the transaction costs of alternative courses of action. These costs must be compared with the potential benefits to be achieved since different water sources and types of agricultural chemical pollution will have different control costs. For example, inducing farmers to reduce their excessive use of nitrogen is likely to be less costly than having them change weed control practices, i.e., reduce the use of herbicides.

Transaction Costs

When designing policies, programs and policy instruments to reduce the level of water pollution by agricultural chemicals, a clear understanding is required of the transaction costs involved in implementing each alternative including search and information costs, bargaining and decision making costs, monitoring and enforcement costs, as well as any litigation costs (Williamson 1985). The distribution of costs and benefits involved with each alternative approach will determine to a large extent, their political support and the level of transaction costs. Ways to reduce such transaction costs need to be explored across alternative policies and policy instruments.

The fundamental unit of analysis will be the transaction which Williamson (1985) defines as something that "occurs when . . . one stage of activity terminates and another begins." In the case of water pollution, transactions occur whenever water is treated, or has wastes dumped in it, or when new agricultural policies or institutional arrangements are developed. Transactions also include changes in farming enterprises or farming practices. The farm plan that SCS develops for farmers is a transaction that involves a contract with farmers that is difficult to enforce and costly to develop.

The transaction costs of principal interest in developing alternative policies, institutional arrangements, and policy instruments for reducing water pollution in agriculture include, 1) the costs of enacting policies and programs, and 2) the costs of their implementation with specific policy instruments and institutional arrangements. The latter involves government costs (monitoring and enforcement costs and administrative and information costs) and must consider compliance costs imposed on farmers and the chemical industry. There will be a feedback between the compliance costs imposed on farmers and the chemical industry and the transaction costs of enacting policies and programs. For example, the transaction cost of promulgating improved water quality (through less use of agricultural chemicals) as a specific objective in the farm bill would likely be high. Farm groups and the chemical industry would strongly, oppose the idea because of their expected loss in income. In contrast, it is likely. to be more difficult to build continued support among environmental groups to offset these increases in transaction costs because their gains are smaller per individual and less clear cut. However, environmental groups have used ideology as a means to reduce the transaction costs of organizing to promote such restrictions (Nabli and Nugent, 1989).

The size of these transaction costs will depend on a number of factors including: asset specificity, - information availability and use, – opportunism – frequency of transactions, — creditable commitments, - uncertainty, and — the characteristics of land and water resources involved.

Information and Opportunism

The assumptions of bounded rationality and opportunism will be particularly important since the benefits and costs of water pollution control will not be uniform across the landscape."'Transaction cost economics pairs the assumption of bounded rationality with a self-interest seeking assumption that makes allowance for guile (opportunism). Specifically economic agents are permitted to disclose information in a selective and distorted manner. Calculated efforts to mislead, disguise, obfuscate and confuse are thus admitted." Transactions, therefore, must be organized "to economize on bounded rationality (limits on information and ability to process it) while simultaneously safeguarding transactions against the hazards of opportunism" (Williamson 1989, p. 12-13). Clearly, changes in the availability of information, the way it is presented and the ability of farmers and government agencies to process it will affect the transaction cost.

[2] Underline added by author.

Information is made more imperfect by the existence of opportunism and the incentives for farmers and/or chemical dealers not to cooperate. For example, what monitoring and enforcement costs of restrictions on chemical use would be required to assure that farmers do not under-report chemical use?

Asset Specificity

The differences in asset specificity across farm types mean that the transaction costs of responding to changes in policies or institutional arrangements will be quite different among farms, e.g. dairy farms as compared to wheat farms. "Asset specificity has reference to the degree to which an asset can be redeployed to alternative uses and by alternative users without sacrifice of productive value . . . It is asset specificity in conjunction with bounded rationality, opportunism and uncertainty that poses the contractional/organizational strains" (Williamson 1989, pp. 13-14). In the case of water quality, they will cause different levels of strain depending on which alternative control strategy is implemented.

Uncertainty and Frequency of Transaction

Along with asset specificity, Williamson (1985) identifies two additional dimensions which make transaction cost economics important in addressing the problems of agricultural pollution of water: 1) uncertainty and 2) frequency of transactions. Uncertainty is critical in both the farming operation and in the control of water pollution because of bounded rationality and opportunism. As uncertainty increases, more information must be processed in making decisions and in implementing decisions which adds to the transaction costs. In response to this uncertainty, investment may have to be made in information systems or in organizational changes at the farm or regulatory agency level (Galbraith, 1973).

The frequency of transactions is important because of the benefits from specialized governance structures or organizational arrangements. "Specialized governance structures are more sensitively attuned to the governance needs of nonstandard transactions than are unspecialized structures, _ceteris paribus_. But specialized structures come at a great cost, and the question is whether the costs "can be justified. . . . The cost of specialized governance structures will be easier to recover for large transactions of a recurring kind (Williamson 1985, p.61). For agricultural chemical pollution of water supplies the key question is whether or not it is possible to use existing agencies like Soil Conservation Service (SCS), Agricultural Stabilization and Conservation Service (ASCS) and the Extension Service to implement the necessary transactions to reduce water pollution. If they cannot or do not have the will to regulate pollution, and EPA or a new specialized agency must do the job, then the transaction cost of controlling water pollution will be substantially higher.

Another important aspect of governance structures or organizational arrangements is that they provide different levels of safeguards, incentives and adaptability. These differences would, therefore, occur across policy instruments and institutional arrangements since they require different types of governance structures. For example, taxes on agricultural chemicals depend on market governance while regulations are based on some form of direct government intervention. Taxes provide monetary incentives to reduce chemical use, but might not be very adaptable to changing conditions. In contrast, regulations do not provide monetary incentives but may be more adaptable to changes. Yet well enforced regulations offer better safeguards against exceeding specified levels of pollution than do taxes, although a combination of taxes and pollution standards offer both good safeguards and incentives.

Creditable Commitments

A final aspect of transaction costs that is likely to be important in the control of agricultural chemical pollution is the idea of creditable commitments or assurance concerning the action of others. For example, what assurance or commitment do we have that farmers will use pesticides according to the directions on the label? Williamson (1989) finds that legal sanctions are severely limited and that creditable commitments are needed because of those limitations. For agricultural chemicals this can be important in at least, three levels. First what creditable commitments need to be established between farmers and the public sector to implement an effective program to reduce agricultural chemical pollution? Second, other sectors of the economy have to make creditable commitments to reduce chemical water pollution so that farmers feel others are doing their fair share, i.e., urban residential and golf course users of chemicals. Third, creditable commitments have to exist among farmers so that they will abide by the rules and limit chemical use. If most other farmers are thought to be cheating, why should they follow the rules? Finally, the same types of creditable commitments need to be established with pesticide and fertilizer dealers. This is particularly important when they apply chemicals and/or are used as the point of regulation or taxation.

Policy Options.

To significantly reduce the level of water pollution by agricultural chemicals will require changes in the farming sector. Figure 1 indicates many of the important linkages in the farming sector, and shows where government policies and programs have an impact on the agricultural sector. These many linkages suggest that to significantly change chemical use in agriculture will require a broad-based approach, starting with trade and agricultural policies and working all the way down to technical assistance provided to farmers by SCS.

We need to be concerned with how trade and agricultural policies influence input use in agriculture. Do they encourage intensive farming and the substitution of agricultural chemicals for land and labor? If so, what changes can be made to reduce or eliminate such incentives? One starting point would be to make reduced agricultural chemical levels in water supplies a specific objective of agricultural policy, and include it in all legislation related to agricultural production.

The next step would be to develop specific policy instruments and institutional arrangements to help achieve this objective. An important aspect of selecting the policy instruments or institutional arrangements is that they are likely to have different degrees of effectiveness depending on whether they are used to reduce surface water or ground water pollution. Since surface water pollution is much more of an externality problem than is ground water pollution, the methods for improving surface water quality should be focused on internalizing the externalities. In contrast, ground water pollution appears to be more an information problem where educational and technical assistance programs should be more effective. Furthermore, there may be some important differences in the spatial variability of chemical pollutants that must be taken into account. For example, is Atrazine contamination more localized than that from nitrates?

Some of the alternative policy instruments and institutional arrangements that should be considered for managing water quality include the following (1) subsidies, technical assistance and education (the traditional approaches), (2) bans on chemical use, (3) taxes and user permits, (4) land retirement, restrictions on chemical use and direct payments and (5) pollution rights and liability. The transaction costs of these alternatives will vary widely because of the institutional and organizational arrangements that already exist in the agricultural sector. Differences in information uncertainty, and asset specificity across regions and farm types, along with the possibility of opportunistic behavior by farmers, creditable commitments and the frequency of transactions, will all have a major affect on the level of transaction costs.

Subsides, Technical Assistance and Education (traditional approaches)

A review of policy instruments suggests some wide differences in transaction costs, particularly in terms of support from the farming sector. Cost-sharing (subsidies), education and technical assistance, to encourage the adoption of best management practices, have been the traditional public sector approaches used in the U.S. to control soil erosion and to reduce nonpoint pollution of surface water (Easter and Cotner, 1982). This is not an accident. These approaches are the most acceptable to farmers because they are free to participate or not and the programs also reduce the farmer's costs of adapting conservation practices. The U.S. also has existing agencies that have experience in providing conservation and pollution control services, i.e., SCS, ASCS and the Extension Service. This combination of existing agencies, no enforcement costs, and farmer support lowers the transaction costs of this set of alternatives particularly in the case of surface water (Table 1). However, the same set of practices and cost-sharing arrangements are not as effective for protecting ground water quality as they have been in reducing soil erosion, although some would argue about their effectiveness in reducing soil erosion.

The subsidy for soil conserving practices is one that tries to reduce pollution by changing the technology used. Another more general type would be a subsidy for meeting a set level of water quality. Farmers could then meet the standard with the lowest cost method which may or may not involve a change in technology (practices). The subsidy based on meeting a given standard requires establishment of baseline water quality and monitoring of water quality which is usually very dependent on rainfall events. Both requirements would substantially raise the transaction cost of reducing agricultural water pollution. Again, this helps explain why the traditional approach is being tried.

If SCS continues to have a major role in helping to reduce water pollution, serious questions need to be asked concerning their basic approach. For example, is a whole farm plan a cost-effective way to control chemical pollution of surface or ground water? The dollars spent on developing farm plans might be better spent on developing new farming practices and promoting their use.[3] Since new approaches are needed, training programs for SCS, ASCS and county extension service personnel maybe critical for program effectiveness. Thus the transaction cost of using the traditional approaches may not be as low as it first appears.

It is likely that best management practices and farming systems to reduce agricultural water pollution will have to be region specific, which will raise the cost of their development. Research will be needed to determine the impact of alternative farming practices and systems on ground water supplies under different resource conditions. Currently the lack of such information limits the effectiveness of cost-sharing, educational and technical assistance efforts in the protection of ground water supplies.

The type of research and education effort that is needed is being conducted in the karst area of southeastern Minnesota. Nitrates were identified as the major agricultural chemical polluting the ground water in this porous soil with numerous sinkholes. Research conducted by Legg, et. al., (1989) showed that excessive applications of nitrogen were being applied mostly by livestock farmers that failed to give adequate credit for manure. Further research now suggests that even recommended rates of nitrogen fertilizer application are too high to optimize profits in corn production. The research also shows the nitrate levels in soil water below the root zone (five feet) increases rapidly as nitrogen applications increase (Figure 3). Educational material showing these relationships are now being used by the Minnesota Extension Service to moderate farmers' use of nitrogen fertilizer and manure.

Bans on Selected Chemicals

The U.S. experience with policy instruments includes bans on selected chemicals that have been identified as particularly damaging, such as DDT. Chemical bans have been quite effective, but it takes time to lower the transaction costs of this alternative by building up political support for enactment of a specific ban (Table 1). We are now at the point where bans on herbicides are being enacted because of herbicide pollution of ground water. Current discussions about bans are focused on Alachlor and Atrazine, both widely used herbicides in the U.S.

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[3] "In a dynamic setting where technology can change, there will be transaction costs involved in gaining access to that technology and inducing the relevant agents to adapt their routines so as to accommodate these changes. Hence in such a setting the distinction between production and transaction costs is likely to be blurred." (Nablo and Nugent, 1989, p. 69.)

An important transaction cost that must be considered when enacting bans or imposing chemical use restrictions is monitoring and enforcement costs. When bans or restrictions on chemical use are imposed, there is a trade-off between farmer compliance and the government's monitoring and enforcement costs. Farmers will tend to exceed chemical bans or use restrictions as long as their expected gains from illegal chemical use exceed their expected losses from government imposed penalties. These expected losses, OL, will be directly related to government monitoring and enforcement expenditures and the level of fines imposed (Figure 4). The marginal loss curve, OL is constructed based on a particular level of monitoring and evaluation expenditures. An increase in monitoring and enforcement expenditures will shift the farmers' marginal loss curve from using illegal chemicals to the right to OL" while a reduction will shift it to the left to OL'. Farmers will apply illegal chemicals up to the point the marginal gains, GO, equal the marginal losses from the expected government imposed penalties. If the farmers' marginal loss curve is OL then they will use OU chemicals (the point where the slope of OL is equal to the slope of GO. The optimum level of monitoring and enforcement is OQ at a cost of 01 given the pollution cost curve AFP (the minimum point on the total cost curve AFP and the point where the marginal cost of monitoring and enforcement equals the marginal pollution cost). The pollution cost curve is constructed from the locus of, equilibrium levels of chemical use given by OCDEN which is constructed from different OL curves.

The curve ART shows the total cost to society from pollution and its control. It is a combination of' monitoring and enforcement costs and pollution costs. Thus the higher the level of pollution costs, the greater the monitoring and enforcement costs that would be economical to use. More monitoring and enforcement would be justified if the pollution cost curve AFP shifts to the right and less if it shifts to the left. Improved monitoring and enforcement technology could also change the minimum cost level. This same relationship would exist between monitoring and enforcement and chemical sales if chemical and fertilizer dealers were regulated. In this case, both dealers and farmers would consider the potential gains and losses from selling and applying excessive chemicals.

If an individual state or nation bans selected herbicides, what might be the impacts on farmers, the input industry and rural communities? One likely possibility is that the impact of a ban on a few selected herbicides would be minor, particularly if there are good substitutes that are less likely to reach "the ground water, i.e., they are less water soluble or break down more quickly. Enforcement would also be less costly because the farmer's gain, GO, would be less from using the illegal chemical. The curve OCDEN would be lower as would the pollution cost curve AFP.

In the case of a ban on Atrazine, the impact on net returns to farmers and gains from noncompliance depends on the weather conditions for weed control (Cox and Easter, 1990). If the weather is good for weed control, substitutes for Atrazine provide satisfactory weed control with only a small decrease in net returns. When the weather is unfavorable for weed control, the decline in weed control and resulting drop in yields can be substantial. The drop in estimated farm net returns for southeastern Minnesota would be around $20 per acre with unfavorable weather (Table 2). Thus the impact of bans and enforcement costs will depend on weather conditions and how much risk farmers are willing to accept when selecting weed control methods.

Bans on Alachlor should have a smaller economic impact on farmers and probably involve lower enforcement costs than those for Atrazine, since there are a number of good substitute herbicides. However, when Alachlor was banned in Canada, the chemical firms exhibited opportunistic behavior and raised the price of the substitutes by over 15%, which significantly increased the cost of weed control. If both Atrazine and Alachlor are banned, the drop in net returns would be somewhat greater than for just Atrazine or Alachlor alone, because of limited substitutes. The loss in net returns to farmers would be even higher if cropping system changes are required to improve weed control, particularly when substantial new capital investments are required and existing capital assets have few alternative uses (high asset specificity and low salvage values). Farm asset fixity or specificity raises the transaction costs of making major changes in farming systems. Thus enforcement costs for a ban on both Atrazine and Alachlor could be high, particularly if it was a state or regional ban.

The ban could also have a differential impact regionally. For example, southeastern Minnesota generally has good rainfall and better weather conditions for a range of different herbicides to be used to control weeds. In contrast, western Minnesota is drier and weather is not as suitable for use of some herbicides. This means that a ban on selected herbicides could cause a greater increase in weed control costs for western Minnesota then it does for the southeast. Because of the dry conditions, farmers might have to shift mostly to mechanical weed control. Thus bans on selected herbicides may put certain regions, such as western Minnesota, at a competitive disadvantage and farmers would have greater incentives not to comply, which could raise enforcement costs.

Government bans on chemical use may take place at an even lower level than a state. Just as individual counties have raised their standards for domestic drinking water, they could also take direct action to ban farming practices that contribute to chemical water pollution. A county might ban certain manure handling practices, or the sale or use of Atrazine. In conjunction with such restrictions, the county could help farmers install manure storage facilities or develop markets for their excess manure. Subsidies for alternative, less polluting herbicides might also be used so the county's farmers are not at a competitive disadvantage to other regions. Such combined actions would help keep the negative financial impacts for farmers to a minimum and help reduce their opposition and the transactions cost of implementing such environmental restrictions. However, with outright herbicide bans, what is to prevent opportunistic farmers from taking their business across the border? This, of course, will not please local businesses and will raise the transaction costs of implementing an effective targeted ban. Thus the opportunistic behavior of farmers and input suppliers along with asset fixity could make the transaction costs high for a targeted herbicide ban particularly if it alters farming systems.

An additional problem arises if the ban is targeted just on areas susceptible to water pollution. The susceptible areas have to be identified, which will increase information costs and raise difficult questions concerning what farms to include in the targeted area. Should everyone with land over aquifer or near a stream be included, or should it be everyone in the county or watershed? Again, opportunistic behavior can be expected from farmers who do not want to be included in the targeted area. Combining this with the information costs suggests high transaction costs.

A final issue involves the impact on consumers of reduced agricultural chemicals. Likely, chemical bans will mean reduced U.S. agricultural production and more food imports. For the consumers' budgets, it would mean higher food prices. Since many agricultural commodities have price inelastic demands, producers will benefit and consumers will lose from higher prices. However, not all producers will benefit, and some will benefit more than the others. This will make the support for drastic restrictions on chemicals somewhat uncertain. Because of the uncertainty over who benefits and who loses, the agricultural sector will, in general, oppose the change, raising transaction costs. Those urban people with moderate to high incomes will probably support restrictions and will be willing to pay somewhat higher food prices for cleaner water. With low income people, the support is less clear cut because of the likely substantial impact of higher food prices on their limited incomes.

Taxes and Permits

The U.S. has had limited experience in using taxes or permits as a means for reducing chemical use. In contrast, Europe has had some success in reducing nitrogen applications through the use of taxes. The problem is that the demand for nitrogen fertilizer may be highly inelastic below certain levels, i.e., 50 to 150 lbs. per acre depending on the soil type, water availability and other factors. A similar situation may exist for certain pesticides. The advantage of taxes is that they can be implemented through fertilizer and pesticide dealers and provide farmers with market incentives to reduce chemical use. This means lower transaction costs in terms of tax collection as well as monitoring and enforcement costs. Dealing directly with each farmer, as would be required with application limits, would greatly increase these costs.

Permits could be used if we knew how much of a chemical is safe to use in a given area. Permits could then be sold or allocated up to the maximum acceptable level of use. One difficulty is that the permitted levels would have to be varied by area, depending on an area's physical characteristics (i.e., soil texture, vegetation and slope) and its location relative to water sources. This information requirement would substantially raise the transaction costs of a permit system. On the positive side, tradeable permits would put a value on the assimilative capacity of agricultural land, and encourage farmers to conserve it. They would also provide an incentive to limit the chemicals used because they could sell unused permits to other farmers. In fact, nonfarmers concerned about water quality could be allowed to buy up permits and reduce the quantity of chemicals applied in an area.

With taxes and tradeable permits farmers may have less incentives for opportunistic behavior and noncompliance than they would with an outright ban since they could legally obtain the chemicals but at a higher price. This should hold down the transaction cost involved with enforcement and those related to fixed assets since there would likely be fewer changes in farming systems than with outright bans.

Land Retirement, Restrictions on Chemical Use and Direct Payments

The U.S., through its farm commodity programs, has made extensive use of land retirement and direct payments to reduce agricultural production and support farm income. Land retirement could be used in the farm program, since it was part of a package which included commodity payments to participating farmers. Would it be possible to include chemical use restrictions as a requirement for participation in the farm programs and what would be the transaction costs of doing so? One major cost would be to get such a provision included as part of the farm bill. Clearly there is a precedent for restrictions on participation in the farm programs with the current requirements concerning soil conservation and wetlands. However, promulgating chemical restrictions, as part of the farm bill, is only one of the transaction costs involved.

The task of implementing a program to retire land or restrict chemical use would probably fall on either ASCS or EPA. In terms of being the most effective (highest will to regulate) in reducing pollution levels, EPA would be the clear choice. On the other hand, ASCS, with the help of SCS, may be the only agency in a position to implement the program since they have a presence in most U.S. counties. The problem is that implementation would require close monitoring and policing, particularly in areas where farmers apply their own chemicals. This, along with the idea that they will be more lenient and sympathetic to farmers, is why ASCS and SCS might be the first choice. Where chemicals are mostly applied by contractors, the control and monitoring could be done through them and transaction costs reduced.

The high transaction costs support the idea that use standards or direct control on the amounts of chemicals applied would work better in controlling agricultural chemical pollution than performance standards (Braden and Lovejoy, 1990). Since chemical use, particularly pesticides use, could be controlled mostly through dealers, the monitoring, enforcement and information costs would be relatively lower. In contrast, performance standards would require monitoring and enforcement at a more micro level which would substantially raise transaction costs. Thus, based just on transaction cost considerations, performance standards are not likely to be a desirable policy instrument (Table 1).

The level of penalties for not complying with chemical restrictions will also have an impact on compliance. If the penalty is a small fine, then compliance is likely to be low without intense monitoring. The farmer's loss, OL, would be low (Figure 4). In contrast, a loss of all farm program benefits because of the illegal use of agricultural chemicals would likely result in higher compliance levels, even without much monitoring. How neighbors respond to the program may also be quite important in determining what an individual farmer will do. If your neighbors support the idea of chemical restrictions and abide by them, then it will likely be more difficult for you to cheat (social or community pressures). There is also the fear and possibility that your neighbors will turn you in if they see you misusing agricultural chemicals or animal **waste.**[4]

A land retirement program and/or easements might be used to protect areas with highly valued water supplies (Figure 2). Programs such as Minnesota's RIM have been used to buy easements for restoring wildlife habitats. Similar programs could be designed to protect valuable ground and surface water supplies. The transaction costs could be high for such a program since the task of determining which water supplies to protect could be highly political. It will also require a lot more information concerning the susceptibility of ground water supplies to chemical pollution and their potential future use.

Land retirement and easements would also be costly in terms of direct payments to landowners. Yet easements would be lower in cost than land retirements if farmers could continue to use the lands as long as they did not apply herbicides. Still, someone would have to enforce such restrictions on herbicide use which, of course, raises the transaction costs. In addition, increasing mechanical weed control could increase soil erosion and augment surface water pollution particularly in steeply sloping areas.

Lovejoy (1990) suggests that SCS and ASCS buy the rights to certain types of erosive land use practices to control soil erosion in erosion prone areas. He further suggests an innovative method for reducing the transaction costs of enforcement, where the property rights are assigned to some group or organization interested in protecting water quality. Organizations interested in protecting the environment such as the Nature Conservancy or Izaak Walton League would be given the partial property rights and if these contractual obligations were violated by farmers, the environmental organization could take judicial action. This, however, is just a transfer of costs and not a reduction in transaction costs for society and could cause over protection.

Since the practices that might be prohibited for soil erosion would have a fairly visible impact on the landscape, monitoring should not be costly. However, if the same approach was tried for agricultural chemicals, more intensive monitoring would be necessary. For example, how do you know that a farmer applied two pounds of Atrazine per acre on a given field, rather than three or four pounds? (Two pounds per acre is the limit on Atrazine use in Wisconsin.)

<u>Pollution Rights and Liability</u>

Implementing changes in property rights regarding water quality is a much broader issue than just agricultural chemicals. It also must include point source water pollution as well as water pollution by soil particles since rights to clean water should

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[4] The freedom and ability to organize and protest against unwanted externalities is one way to prevent excessive pollution. The lack of such freedom may explain why pollution got so bad in Eastern Europe. This same freedom and ability to organize locally in many areas of the U.S. could be used as a means to reduce the transaction costs of controlling agricultural pollution through bans or regulations.

involve all sources of contamination. The idea that the citizens of the U.S. have the right to clean water has been legislated in the U.S. clean water act. The problem is putting such objectives into practice. There are limits on the amounts of pollutants that point sources are allowed to deposit in lakes, rivers and streams, however, the rights to "clean" water do not exist de jure. This is particularly true in terms of nonpoint sources and a number of point sources. The transaction costs of implementing such a major change in water quality rights is high, particularly in the short run (Table 1). In fact, in the short run, it is almost impossible to implement because of the chemicals already in the soil or stream beds that will eventually enter our water supplies without any additional discharges or applications of chemicals.

One means of moving towards a policy of giving the U.S. citizens the property right to clean water would be to change the liability rules for water pollution. Polluters could be made liable for any damages or loss in uses caused by their pollution of water. For example, in Connecticut, liability has been imposed on individuals (including farmers) shown to have contaminated drinking water sources. This shifts enforcement to the court systems and, if strictly enforced, could produce some major changes in farming behavior. The major problem is being able to show or prove a farmer has polluted a particular water source while others have not.

The liability for water pollution could also be placed on agricultural chemical manufacturers or dealers. This would act as a tax on farmers because the manufacturers would have to charge a high enough price to cover the liability costs.[5] Manufacturer liability would work just as well as farmer liability except where nonpurchased inputs (manure) are used or where the methods and timing of application by farmers affect pollution rates (Braden and Lovejoy, 1990, p. 50-53). Thus, in areas where livestock production is important and/or farmers apply their own pesticides, a farmer-based liability may be necessary.

Thus, an important first step in making such a rule change effective would be to collect adequate information so that the polluters could be identified and the damages estimated. This would be a major monitoring cost for some types of pollutants because of the temporal and spatial nature of their damages. Another transaction cost is the litigation costs that would be imposed on an already overburdened court system which is not a small cost. For example, Kopp, et. al., (1990) point out that as much as "30 to 70 percent of all current expenditures related to Superfund take the form of legal fees, as opposed to expenditures for actual removal or stabilization of hazardous substances at waste disposal sites" (p. 13). Possibly court costs could be reduced or eliminated through bargaining to obtain out of court settlements which could benefit all participants. In fact, clearly established liabilities should encourage bargaining solutions if only a few parties are involved. However, the threat of court battles will not guarantee that the negotiated outcome is economically efficient (Porter, 1988.)

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[5] Negligence is like a regulatory standard where the firm has no incentive to do better than the safety standard. For the liability rule, there is always some incentive to do better since this will further reduce the chance of liability from pollution.

An alternative approach would be to take action at the county government level through county commissions and land use planning efforts. For example, in Fillmore County Minnesota, a farmer with excess soil erosion can be required to implement a soil erosion control program approved by SCS and enforced at the township level. Similar restrictions could be used by counties to protect their water supplies against chemical pollution. Enforcing restrictions or liabilities at the local level would reduce the transaction costs because farmers tend to know what their neighbors are or are not doing, which could reduce information costs. These local efforts would be most effective where the pollution affects a substantial number of county residents other than the individual farmer, i.e., there is a large negative externality. Highly visible erosion and pollution such as gully erosion, muddy streams and murky lakes are good targets for local action. People can see the damages and are willing to put pressure on county commissioners to take action.

Enforcement and monitoring costs may also be lower because of the social closeness of people in the rural community. When those causing the water pollution are well known in the community, social pressures, obligations and respect for neighbors will influence farming decisions (Robinson and Schmid, 1989). The greater this sense of social closeness the less likely a farmer is, knowingly, going to create a negative water pollution externality In fact, social closeness can be sufficient to maintain negative production externalities at social optimal levels. Such a level would be reached when the cost to the polluter of reducing the negative externality would equal the increased utility received by the pollutee. Community education concerning the impacts of farming on water quality could be an important policy instrument that would complement local attempts to reduce water pollution. As Braden (1990) suggests “a sense of obligation may be transferred with the knowledge that one’s action substantially affect other people” (p. 27). This sense of obligation will even be stronger with social closeness.

Difficulties arise- with county-specific regulations because of the fear that they may put the county’s farmers at a competitive disadvantage and also because chemical water pollution is not visible and crosses county and state boundaries. This is why state or national standards and pollution control efforts that focus on the watershed or aquifer are important. Such approaches can internalize many of “the externalities that cross political boundaries.

Strategy to Reduce Agricultural Chemicals in Water Supplies

Because of past levels of agricultural pollution, implementing an effective clean water policy for agriculture requires a long run point of view. It means cutting back on chemical use in agriculture, a much greater use of alternative farming practices, and

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[6] The lack of social closeness had a lot to do with the soil erosion restriction imposed in Fillmore County. An increased number of absentee landowners who employed outside management to run their farms was a major concern of Fillmore County Officials. They felt that these “outsiders” were operating with very short time horizons and that excessive soil erosion was taking place. Since these people were outsiders, social pressures were not effective in inducing them to reduce their erosion externalities. Thus more formal means where found to limit the erosion.

reductions in lawn chemical use in cities and towns. Farmers are not going to cooperate if they feel others are not making “credible commitments” to the reduction in chemical use (Williamson 1985). In addition, the effect of such cutbacks may be limited at first because of the chemicals that already exist in our soil and water resources. A first step would be technical assistance and education, with demonstrations concerning what can be achieved with fewer chemicals applied more often, but in smaller amounts. Use of fewer chemicals in smaller quantities will require more labor and better management skills to maintain production levels. Moderate sized farms may have an advantage over large or corporate farms because of the importance of timing in areas dependent on rainfall. Irrigated areas not dependent on rainfall during the crop season may also have an advantage over nonirrigated areas because control over water reduces the uncertainty involved in weed control and fertilizer use.

The real question is what mix of policies and policy instruments has the best chance of reducing agricultural chemicals in our water supplies over the long run. The whole process of reducing chemical use in agriculture would be facilitated if it was a goal of U.S. farm policy. Such a national goal would lower the transaction costs of taking action at the state and county levels. As a start, local variation should be allowed because of the wide differences amongst regions in terms of physical and climatic conditions, crops grown and inputs used. Experimentation should be allowed, since we still have a great deal to learn about the effects of reduced chemical use and how the chemicals can best be kept out of water supplies. Experimentation is also needed with rule making for monitoring and enforcing agricultural pollution control. If rules are flexible, innovative ways can be developed that reduce transaction costs.

A strategy involving technical assistance, education and cost-sharing for best management practices is favored by the existence of agencies which provide these services. Currently this strategy is being tried on a limited scale for ground water protection and should be evaluated for its cost-effectiveness. Other alternatives should be tested, including bans, use permits and easements in sensitive areas. The education effort should not be limited just to farmers, but should involve the broader population so that they have a better understanding of the problem. This has at least two possible benefits: first, an informed population will be more willing to pay higher food prices resulting from reduced chemical use and second, the nonfarm population can apply community pressure on farmers to limit their chemical uses.

Conclusion

Although the extent of agricultural chemicals in U.S. water resources is still a matter of debate, most individuals would agree that it is a problem for many areas with intensive crop and/or livestock agriculture. The question is, what can and should be done about it? First we should quickly expand our research effort so that we have a better information and knowledge base from which to design our strategies for reducing agricultural chemical water pollution and reduce the transaction costs of implementing different strategies. Second, we need to improve the information available to farmers, and its transmission to farmers, concerning how “best” to use agricultural chemicals and animal wastes while minimizing their negative impacts on water supplies. As part of this, demonstrations of different low input agricultural strategies should be developed throughout the U.S. Cost-sharing arrangements should be tried for system changes that involve high asset fixity and, therefore, high transaction costs. Third, if education and technical assistance along with cost-sharing are not effective then more coercive instruments will have to be used. For example, the liability rule could be changed so that farmers are liable for their water pollution damages. User permits and taxes should also be tried.

Finally, a broad based educational program is needed for the general public so that they can make “better” informed decisions concerning water pollution. For example, what chemical levels pose real risks to humans? In addition, why is it alright to have different chemical levels in the water supply, depending how the water will be used in the future and the assimilative capacity of the water resources? Nitrates in drinking water can cause adverse effects on humans and livestock, but in irrigation water, it can increase crop yields and lower fertilizer costs.

Transaction costs play a major role in determining the U.S. strategy for managing agricultural chemical use. This is why the President’s Water Quality Initiative emphasizes the traditional approaches, such as technical assistance, education and cost-sharing, which are implemented by existing agencies. If these efforts are not successful, there will be increased pressure to try more coercive control measures with correspondingly higher transaction costs. This is when farmer compliance with alternative pollution control instruments will become critical and determine the level of monitoring and enforcement costs that will be necessary to achieve water quality goals. A noncooperative rural community could mean that monitoring and enforcement costs are prohibitively high. In additional, a strongly opposed rural community could raise the transaction costs so high that passage of any effective legislation to reduce agricultural chemical use would be blocked.

As economists, we need to estimate the transaction costs for alternative approaches to reduce agricultural water pollution and help design institutional and organizational arrangements that will reduce transaction costs. For example, can arrangements be designed that channel community concerns towards effective local and state based efforts to reduce agricultural chemical pollution of ground water? Farmer response and the transaction costs of reducing chemical use will not be uniform across the United States, or even across an individual state. Consequently, community based approaches might be the most cost effective approach, particularly when water pollution impacts are mostly localized, i.e. ground water pollution. However, when the problem crosses state or county boundaries, these local efforts are not likely to be enough. In addition, when other concerns such as economic development dampen local interests in reducing water pollution, then the federal government may have to step in to prevent or reduce water pollution.

Of course, decisions will have to be made before all the information we would like is available. The Canadian ban on Alachlor is an example of one such decision. Hopefully, the U.S. can approach the problems of reduced agricultural chemical water pollution in a more systematic and targeted fashion.

TABLE 1. THE TRANSACTION COSTS OF ALTERNATIVE POLICY INSTRUMENTS

Policy Instruments	Transaction Costs					
	Search and Information	Bargaining and Decision Making	Monitoring and Enforcement	Litigation	Compliance Costs	Program Effectiveness in Controlling Pollution
Traditional Approach (cost-sharing, technical assistance & education	moderate	low	low	none	low	low
Chemical Bans						
1. National	moderate	high	low	low	high	high
2. Local	high	moderate	high	low	high	moderate
Taxes	moderate	high	low	low	low	low
Permits	high	high	moderate	low	moderate	high
Land Retirement	moderate	high	moderate	low	moderate	moderate
Easements	moderate	high	high	low	moderate	moderate
Standards						
1. Performance	high	high	high	moderate	high	high
2. Use or Practice	moderate	moderate	low	low	low	moderate
Property Rights and Liability	high	high	high	high	high	high

Table 2. CHANGES IN NET RETURNS DUE TO HERBICIDE BANS ON SOUTHEASTERN MINNESOTA FARMS USING CONVENTIONAL TILLAGE PRACTICES

TYPE OF BAN & DECISION RULE	TYPE OF WEATHER THAT OCCURS	
	GOOD	BAD
	--------------(per acre)-------------	
BAN ATRAZINE		
Maximum Net Returns, Assuming Good Weather	-$0.51(0%)	-$20.50(10%)
Maximum Net Returns, Assuming Bad Weather	-$7.73(3%)	-$7.73(4%)
No Herbicide	-$11.62(4%)	-$71.76(35%)
BAN ALACHLOR		
Maximum Net Returns, Assuming Good Weather	-$0.10(0%)	-$20.15(10%)
Maximum Net Returns, Assuming Bad Weather	-$2.64(1%)	-$2.64(1%)
BAN ATRAZINE AND ALACHLOR		
Maximum Net Returns, Assuming Good Weather	-$0.51(0%)	-$20.56(10%)
Maximum Net Returns, Assuming Bad Weather	-$9.53(3%)	-$9.53(5%)

Source: Craig A. Cox and K. William Easter, 1990.

Figure 1. Agriculture Related Water Quality System

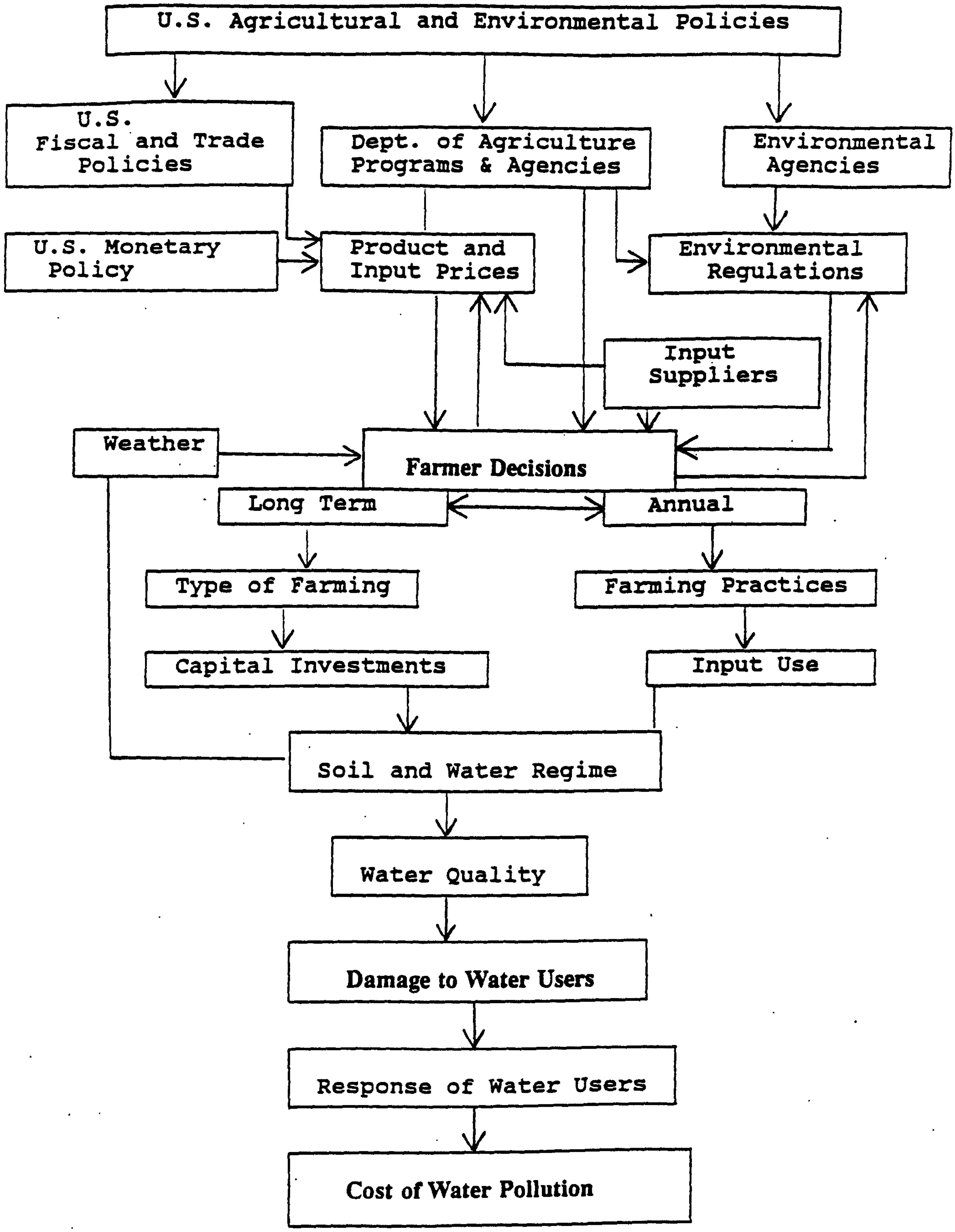


Figure 2. Relative Size of Benefits from Controlling Agricultural Chemical Pollution of Water Supplies

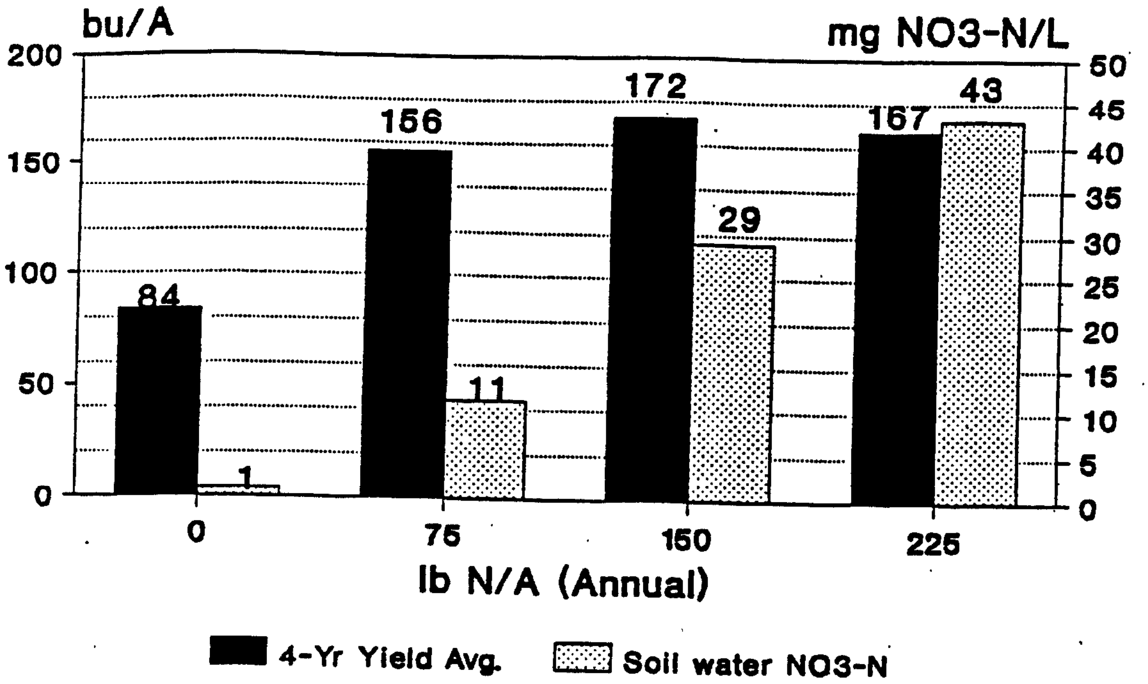
Non-point pollution
Susceptibility of Water Source

Value of Water Use[1]	Low	Medium	High	Cost of Alternative Water Supplies[2]
Low				Low
Medium				Medium
High			High Benefits	High

[1] This is a function of water use in the region and the growth in population, income and the water using sectors of the economy. Irrigation water uses would generally have a low value while industrial and domestic consumptive uses would have a high value.

[2] Includes cost of clean up as an alternative.

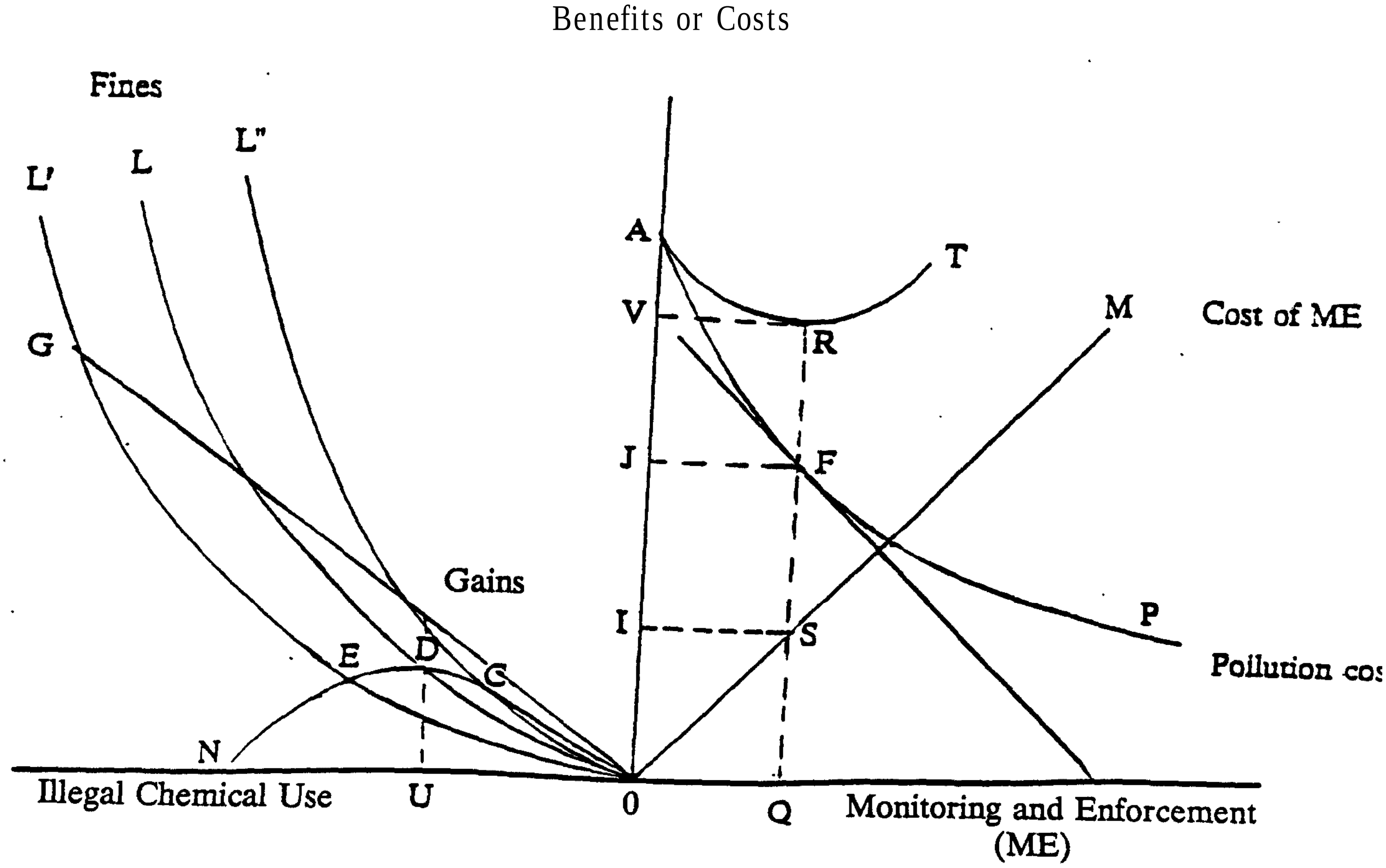
Figure 3 Relationship between Yield and Nitrate-N in soil water at 5' in September 1990.



Lawler Farm, Olmsted County, Minnesota.

Source: Randall, et. al., 1991.

Figure 4 Monitoring and Enforcement Cost of Restriction on Agricultural Chemicals Used.



Source: Adapted from Nabli and Nugent, 1989, p. 48.

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